



THE FATE OF DISCARDS **FROM MARINE FISHERIES**

**A disregarded viewpoint in
fisheries management**

JOCHEN DEPESTELE



THE FATE OF DISCARDS FROM MARINE FISHERIES

A disregarded viewpoint in fisheries management

DE BESTEMMING VAN TERUGGOOI IN ZEEVISSERIJ

Een blinde vlek in het visserijbeheer

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ISBN : 978-90-5989-816-5

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Academic year 2014-2015

Publically defended on 18 September 2015

This thesis is submitted in partial fulfilment of the requirements for the degree of Doctor of Science in Biology

Please cite this thesis as:

Depestele, J. 2015, The fate of discards from marine fisheries. PhD thesis Ghent University. Gent, Belgium. 286pp.

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Summary

Fishing at sea has intensified since the industrial revolution in the 20th century, not only by the development of wide range of fishing gears with increasing catch efficiencies, but also by the spatial extent of the fishing grounds, now covering vast areas from coastal regions to the deep-sea. It was, however, not until the second half of last century that the study of ecological effects of fishing was directed towards a broader spectrum than the fished populations themselves. The interactions in the marine ecosystem are bewilderingly complex, even without considering fishing disturbances. Fisheries' science gradually learns that these ecosystem interactions are of great concern if commercial populations are to be sustainably exploited. Moving from this global notion towards concrete integrated ecosystem assessments remains highly challenging though. The majority of the measures of European fisheries management remains focused on isolated topics, such as harvest strategies of single species. The return of unwanted catches, defined as discards, is at the centre of latest Reform of the Common Fisheries Policy. The thesis contributes to bridging the gap between discards as a single species population effect towards the understanding of its effects at ecosystem level.

The thesis aims to work towards the quantification of the fate of discards as seen from the human (or stock) perspective as well as the ecosystem perspective. Partitioning the fate of discards should provide the building blocks to assess the significance that discards make (or not) to the ecosystem, and how the ecosystem structure and function is altered by discarding.

The human perspective on discards was mainly linked to the amount and composition of the discards produced, and the possibilities for discard reduction. Discard management requires reliable estimates of the magnitude of the discards in several circumstances. The sampling coverages of most European discard observer programmes are low, and require a different approach if we are to understand the variability of discarding across regions, fleets, periods, species, and so on. The causes of discard variability were investigated for four commercial fish species, common sole (*Solea solea*), European plaice (*Pleuronectes platessa*), Atlantic cod (*Gadus morhua*) and whiting (*Merlangius merlangus*) in the Belgian beam trawl fishery in the southern North Sea. Size composition was the main driver for sole and plaice discards, but the discards of whiting and cod also comprised marketable fish. The possibility to estimate discards from length-frequency distribution as an additional source of information may work partially, but still requires increased understanding of other drivers of discarding that could not be explained by the case study. Other possibilities to

increase the sampling coverage were discussed, and included a self-sampling system scheme whereby fishermen sample the catch themselves, and a system of electronic monitoring devices using cameras. These systems also focused on commercial fish species, but may not work in all circumstances. Possibilities to monitor the discards of non-commercial species was primarily realised through modelling studies, and discard observer programmes of isolated member states of the EU. Monitoring discards with a 'sufficient' sampling and species coverage remains a daunting task.

The effects of discarding on an organism were further investigated by an evaluation of the potential of the discarded organisms, both commercial fish and non-commercial invertebrate species, to survive the catching and discarding process. The survival of discarded fish is a topic of hot debate in relation to the landing obligation, which prohibits to discard certain quota-regulated species. An exemption from the landing obligation may be obtained if scientific support demonstrates that the survival of discarded fish is 'high'. The scientific estimates of short-term discard survival, however, are highly variable depending on the species, fishing operations and environmental circumstances. In a first instance, this study served to increase the limited number of empirical data on discard survival. The selected case study was directed towards the 'eurocutter' fishery in the southern North Sea, equipped with 4m beam trawl and chain mats. The survival of sole was ~14% after 91 h of observation, whereas 48% of plaice survived after 77 h of monitoring, 66% of cod (88 h) and 72% of Rajidae (80 h). Whiting and pouting (*Trisopterus* sp.) did not survive the process, while the short-term survival of benthic invertebrate species was >75%. The survival estimates may not be readily extrapolated to estimate survival of discards at fleet level, given the high variability among studies. A survival proxy, an easy-to-measure property with a good predictive power of short-term discard survival, was investigated, but did not result in a sufficient predictive power for all species. The proxy was based on exterior physical injuries and may require additional information on the stress level experienced by discarded organisms, as is accounted for in the Reflex Action Mortality Predictor (RAMP). The potential to develop reflexes for sole and plaice was investigated.

The potential of discards to survive the capture process in the short term does not prevent them from being easy targets for predation. Discards are a substantial food subsidy to several scavenging seabird species. The discards that are consumed by seabirds are extracted from the marine ecosystem and cannot become available to scavengers in the sea. Estimates of the contribution of discards to the food requirements of marine scavengers on the seabed vary from negligible to substantial. The variability may be related to the amount and composition of the discards that becomes available demersal and pelagic scavengers. The thesis developed a framework to account for the spatial and temporal variation in seabird distribution, attraction and discard distribution in order to estimate discard consumption by seabirds over space and time. The framework was applied

to the French fisheries in the Bay of Biscay and showed high variation in discard consumption across seabird foraging guilds, discard types, semesters and locations. The Bay of Biscay case-study showed that seabirds consumed about one quarter of all discards, and that the remainder sinks to benthic communities.

The variability in discard partitioning in space and time indicates that the potential of discards to significantly affect the diet of benthic communities will differ in space and time. The main epibenthic scavengers were identified from commercial trap fisheries, as well as a literature overview from experimental studies of discard consumption by benthic scavenging communities. These species may profit from the additional food source that fisheries generate, depending on their attractivity and the area over which they can be attracted.

Scavenging benthic species may be directly of interest to fisheries, such as *Nephrops norvegicus*, or may be a food source for commercially exploited species. Discards may affect various scavenging species of different ecosystem components. The extent to which this effect may result in a shift of the ecosystem structure and function is not clear as yet, but this thesis has provided the building blocks to move towards food web studies including discards.

Although it is not clear for specific species or species groups how discarding may influence them, the general influence is expected to be substantial (e.g. high mortality rates of whiting, locally subsidised benthic communities, extra food for birds). Measures for the reduction of discards have been proposed, primarily from a fishing gear perspective. A wide range of net modifications were investigated for the Belgian beam trawl fisheries. There may be a potential reduction in the discard levels of benthic invertebrates and certain roundfish species. Yet, the uptake of these measures by the fishing industry has hitherto been largely lacking. Whether the reformed landing obligation will meet its goal and provide sufficient incentives for fishermen to use those gear modifications is to be discovered over the coming years. The lacking implementation of gear measures in Belgian flatfish-targeted beam trawl fisheries prevents fleet-wide evaluation of their efficacy. Parallels with other fishing industries, however, highlight that the efficacy of gear measures vary substantially across individual fishermen, fleets, environmental circumstances, gear riggings, and so on. The envisaged discard reduction will likely require other, complementary measures, such as real-time spatial management of fisheries' catches.

A complete reduction of fisheries' discards may not be possible. Their role as a source of mortality as well as food across the wide range of species and trophic levels requires in the marine ecosystem further observation and analysis if fisheries are to be managed sustainably. The foundations to this end are provided in the thesis.

Samenvatting

De late ontwikkelingen van de industriële revolutie hebben de zeevisserij drastisch gewijzigd. De omschakeling van zeil- naar stoomschepen en later naar motorvaartuigen, het gebruik van vistuigen met een hogere vangstefficiëntie en de uitbreiding van de visgronden van de kust tot zowat alle mogelijke locaties, inclusief de diepzee hebben aangetoond dat de zee niet onuitputtelijk was. Het duurde evenwel tot midden de tweede helft van vorige eeuw eer visserij-effecten bestudeerd werden die zich niet beperkten tot de beviste populaties alleen. Het marien ecosysteem omhelst een complex geheel aan interacties die zelfs zonder visserij-effecten moeilijk te begrijpen zijn. Wetenschappelijk onderzoek toont stap voor stap aan dat het negeren van de interacties tussen bijvoorbeeld vispopulaties onderling onmogelijk kan leiden tot het duurzaam gebruik van mariene hulpbronnen. Uiteraard is de omschakeling van dit besef tot een werkelijk geïntegreerd ecosysteembeheer een enorme uitdaging. Het huidige visserijbeleid richt zich dan ook voornamelijk op de gevestigde beheersystemen voor individuele vispopulaties met slechts een minieme knipoog naar de toekomst om ecosysteemeffecten te integreren. Het vangen en weer overboord zetten van ongewenste vangsten, teruggooi of '*discards*' genoemd, vormt de kern van het huidige Europese gemeenschappelijk visserijbeleid. Deze thesis tracht een brug te slaan tussen het beheer van teruggooi als effect op individuele populaties en het begrijpen van de effecten van teruggooi in het ecosysteem.

In deze thesis wordt er daarom naar gestreefd om de bestemming van de teruggooi te kwantificeren. Het kwantificeren is opgedeeld in verschillende doelstellingen, vertrekkend vanuit het menselijke (*bestands-*) perspectief en gaande tot het ecosysteemperspectief. De bestemming opdelen tussen deze niveaus heeft tot ultieme streefdoel om de bouwstenen aan te leveren voor een algehele evaluatie van de bijdrage van teruggooi aan het ecosysteem, meer bepaald hoe de structuur en het functioneren wijzigt door de extra sterfte (-) of het extra voedsel (+) dat teruggooi oplevert.

Vanuit het oogpunt van de mens wordt vooral geïnvesteerd op een goed beheer van de teruggooi. Hiervoor zijn nauwkeurige data nodig die de variabiliteit precies omschrijven. Hoeveel er wordt teruggegooid, welke soorten of soortengroepen en wat de mogelijkheden zijn om de teruggooi te verminderen, hangt van een waaier van factoren af. De visgronden, de vissersvloten en hun type vistuig, de periode van het vissen en de doelsoorten van de visserij zijn de meest voor de hand liggende. Het inschatten van de variabiliteit van de teruggooi is daarentegen absoluut geen sinecure. De staalnames zijn duur en logistiek veeleisend, wat meestal resulteert in staalname met een lage bedekkingsgraad van de vloot en dus een beperkt aantal kwalitatieve data.

In deze thesis zijn de oorzaken van de variabiliteit van de teruggooi onderzocht voor vier belangrijke commerciële vissoorten in de Belgische boomkorvisserij in de zuidelijke Noordzee: tong (*Solea solea*), schol (*Pleuronectes platessa*), kabeljauw (*Gadus morhua*) en wijting (*Merlangius merlangus*). De voornaamste, zo niet enige oorzaak voor de teruggooi van tong en schol (de doelsoorten) was hun lengte, terwijl van wijting en kabeljauw ook marktwaardige vis werd teruggegooid. De mogelijkheid om de teruggooi op vlootniveau te kwantificeren door gebruik te maken van de lengteverdeling van vis is dus beperkt tot goeie inschatting voor de doelsoorten.

Aangezien het programma met de zeegaande waarnemers te beperkt was om de teruggooi van vele soorten kwalitatief te karteren, blijft de noodzaak om andere mogelijkheden te vinden om teruggooi te bemonsteren. Die mogelijkheden variëren van een systeem waarbij vissers zelf hun teruggooi in kaart brengen tot een alziend oog in de vorm van camera's aan boord. Waar deze systemen de bedekkingsgraad van de staalname verhogen, houden zij andere problemen in, zoals het beperkt aantal soorten dat kan worden bemonsterd. De teruggooi van niet-commerciële soorten kan gedeeltelijk opgevangen worden door modelleerstudies en in enkele Europese lidstaten tevens door een intensief programma van waarnemers. Het in kaart brengen van de teruggooi in de Belgische vloot blijft dus nog steeds grotendeels een afschrikwekkende opdracht.

Het effects van het vissen en teruggooien op de organismen zelf werd geïnterpreteerd als de mogelijkheid van die soorten om het vangst- en teruggooiproces te overleven. De overleving van teruggegooide organismen is een heet hangijzer voor bepaalde vissoorten, en met name voor de soorten die onder de aanlandingsverplichting niet meer zullen teruggegooid mogen worden. Dit verbod zou namelijk kunnen opgeheven worden als het hervormde Europese beleid een uitzonderingsmaatregel toelaat. Een 'hoge' overleving op basis van wetenschappelijke gronden ligt hiervoor aan de basis. Tot nog toe bleken de wetenschappelijke gegevens echter heel variabel volgens de soort, het vistuig en haar toepassing en de milieu-omstandigheden. In deze thesis is vooral gewerkt aan dit hiaat door de overleving te bestuderen van enkele soorten die door de 'eurokotter' visserij in de zuidelijke Noordzee worden teruggegooid. Korte-termijnoverleving werd bestudeerd na vangst door een 4 m boomkor met kettingmat. De overleving van tong was ~14% na 91 u observatie, terwijl de voornaamste teruggooi-soort (schol) tot 48% overleefde na 77 u. Van kabeljauw overleefde 66% (88 u) en van roggensorten 72% (80 u). Wijting en bolken (*Trisopterus* sp.) overleefden het vangstproces niet, terwijl >75% van de benthische invertebraten overleefde. De schattingen kunnen niet rechtstreeks doorgetrokken worden naar de volledige vloot omwille van de variatie aan types visserij binnen de eurokottervloot, inclusief locatie, seizoen, enz. Daarom werd een gemakkelijk te meten indicator uitgetest. Als deze gemakkelijk te meten indicator overleving goed kan voorspellen, dan kan hij geregistreerd worden in meerdere omstandigheden en zo de variabiliteit

van overleving voldoende karteren. De indicator meette de uitwendige schade na het vangstproces, maar bleek slechts gedeeltelijk te korte-termijnoverleving te kunnen voorspellen. Informatie over inwendige beschadiging en stress zou de indicator kunnen aanvullen. Daarom is er onderzocht welke reflexen voor tong en schol zouden kunnen werken. De overleving van vis varieert sterk, en noodzaakt verder experimenteel werk vooraleer kan beslist worden wanneer overleving 'hoog' is op vlootniveau. De indicator voor uitwendige schade en de reflexenmethode zullen hierbij helpen.

De studie van de overleving van teruggewoide vis richt zich enkel op het potentieel om op korte termijn te overleven, en negeert het verminderen van de overlevingskansen door predatie of andere gevaren in het natuurlijk milieu. Zeevogels zouden bijvoorbeeld de overleving van rondvissoorten sterk kunnen hypothekeren. Teruggewoide organismen vormen een wezenlijke bijdrage tot het dieet van aasetende zeevogels. Deze organismen worden grotendeels van de zee onttrokken en kunnen dus tevens niet dienen als voedselbron voor andere, mariene aaseters in de zee. Het belang van teruggewooi varieert bijvoorbeeld voor de benthische gemeenschap van verwaarloosbaar laag tot substantieel hoog. In deze thesis werd het raamwerk ontwikkeld om de ruimtelijke en temporele spreiding van zeevogels, hun aantrekking tot vissersvaartuigen en van teruggewooi in kaart te brengen. Deze factoren werden aan elkaar gekoppeld en geven een inschatting van de hoeveelheid en de samenstelling van de teruggewoide organismen die door vogels worden geconsumeerd en dus niet meer ter beschikking komen van organismen onder water. Dit kader kon worden toegepast voor de Franse zeevisserij in de Golf van Biskaje, omdat alle teruggewoide organismen door de Fransen worden bemonsterd, wat niet het geval is voor de Belgische visserij. De spreiding van de hoeveelheid geconsumeerde teruggewooi was groot en afhankelijk van de zeevogelsoorten, het type teruggewooi, seizoenen en gebied. Dit voorbeeld toonde aan dat zeevogels tot één vierde van de teruggewooi extraheren en met name de voedselrijke rondvis.

De verdeling van teruggewooi tussen aaseters in de lucht en in het water geeft aan dat de biomassa aan teruggewoide organismen het voedselaanbod van benthische gemeenschappen kan aanvullen, en dat deze invloed afhankelijk van locatie en tijd behoorlijk veel kan zijn. De belangrijkste epibenthische aaseters werden geïdentificeerd op basis van commerciële visserijen met visvallen en een literatuuroverzicht van experimentele studies die 'teruggewooi' aangeboden hebben aan de benthische gemeenschap. Hoewel deze bronnen slechts een afwijkend beeld geven van wat er zich effectief op de zeebodem afspeelt, geven ze toch aan dat bepaalde soorten steeds opnieuw van de teruggewooi profiteren. Teruggewooi kan voor die soorten van belang zijn, afhankelijk van de snelheid waarmee ze teruggewooi traceren en afhankelijk van hun competitors. Sommige visserijen zijn gericht op deze aaseters en kunnen een rechtstreeks voordeel ondervinden van teruggewooi, terwijl andere aaseters kunnen zorgen voor een nadelige wijziging in de voedselketen.

Teruggooi heeft een invloed op verschillende trofische niveaus. De mate waarop deze voedselbron een wezenlijke invloed kan uitoefenen op de structuur en het functioneren van het ecosysteem blijft nog onbekend. De eerste bouwstenen om hiertoe te komen werden in deze thesis gelegd.

Hoewel niet meteen ontrafeld is hoe de teruggooi zich precies doorheen het volledige voedselweb voortplant, is een algemene invloed van teruggooi op het ecosysteem niet te ontkennen. Hierbij kan gedacht worden aan voorbeelden als de hoge sterfte van teruggegooide wijting, de lokale beïnvloeding van benthische voedselwebben en de extra voedselbron voor vogels. Het vermijden of verminderen van teruggooi is voor de visbestanden wenselijk en werd voor de Belgische visserij vooral onderzocht vanuit het standpunt van het vistuig. Een breed spectrum aan onderzochte netaanpassingen resulteerde in potentieel voor het verminderen van de teruggooi van rondvis en benthische invertebraten. Afgezien van het gebruik van twee aanpassingen zijn deze mogelijkheden onderbenut in de commerciële visserspraktijken. Eén van de doelstellingen van de hervorming van het visserijbeleid is net om teruggooi te verminderen en daarbij wordt er tevens op gemikt dat ook netaanpassingen zullen worden toegepast. Het uitblijven van het gebruik hiervan in de visserijsector betekent dat de efficiëntie van maatregelen enkel op experimentele schaal is vastgesteld. Naar analogie met andere visserijen zou het kunnen dat de toepassing van deze methodes op vlootniveau niet het gewenste effect behalen. Net als andere maatregelen zullen visserijtechnische aanpassingen moeten nauwgezet opgevolgd worden, en al dan niet aangevuld met complementaire aanpassingen zoals het *real-time* ruimtelijk beheer van visserijen om de vangstmogelijkheden beter te koppelen aan de eigenlijke vangsten.

In deze thesis wordt er op gedrukt dat het volledig verdwijnen van teruggooi heel onwaarschijnlijk is. Omdat teruggooi de ecologie van verschillende soorten aasetende zeevogelpopulaties sterk beïnvloed heeft, en wellicht ook een belangrijke rol speelde bij aaseters van andere ecosysteemcomponenten, zoals benthische invertebraten op de zeebodem, is het belangrijk dat zowel de sterfte als het extra voedselaanbod van teruggooi verder in kaart wordt gebracht en dat haar specifieke rol in het voedselweb wordt verduidelijkt. Hiervoor kan verder gebouwd worden op de funderingen van deze thesis.

Populariserende samenvatting

Het aan banden leggen van de teruggooipraktijken in de zeevisserij zal op termijn zorgen voor minder sterfte van zeedieren, maar kan ook belangrijke verschuivingen teweeg brengen binnen het mariene voedselweb. Het is essentieel dat we begrijpen wat teruggooi betekent voor het ecosysteem, en dat die informatie wordt ingebouwd in het visserijbeleid.

Het terugschroeven van de teruggooi via technische aanpassingen en beheersmaatregelen wordt aanzien als één van de belangrijkste stappen naar de verduurzaming van de visserij. Maar laten we realistisch blijven, teruggooi zal wellicht nooit helemaal verdwijnen uit de zeevisserij. Daarom is het belangrijk dit complex gegeven goed in kaart te brengen: de samenstelling en omvang van de teruggooi, maar ook de overlevingskansen van de dieren die het vangst en teruggooi-proces aan de lijve ondervinden. Sommige dieren worden levend overboord gezet maar dan gretig opgevisst door zeevogels, terwijl andere soorten dood naar de bodem zinken en een belangrijke voedselbron vormen voor aaseters. Via deze thesis is het lot van de teruggooi binnen de visserij beter in beeld gebracht en wordt de broodnodige informatie geleverd voor de verdere ontwikkeling van het visserijbeleid.

Waarom is teruggooi belangrijk?

Bij het vissen met sleepnetten en ander vistuig worden er naast grote, eetbare vissen, ook kleine visjes, zeesterren, schaaldieren en andere commercieel oninteressante dieren gevangen, die na het sorteren van de vangst worden teruggegooid. Teruggooi van ongewenste vangsten is dus een inherent deel van de huidige visserijpraktijk en wordt algemeen aanzien als een vorm van verspilling die de verduurzaming van de visserij in de weg staat. Het aandeel van de teruggooi binnen een vangst varieert sterk tussen vissoorten, types visserij, gebieden en seizoenen, maar kan oplopen tot meer dan 50% in aantallen en in gewicht. Bijgevolg worden er binnen het visserijbeleid grote inspanningen geleverd om de teruggooi in kaart te brengen en terug te dringen. Zo voerde Europa recent de aanlandingsverplichting in. Volgens deze aanlandingsverplichting moeten alle vangsten van quota-gereguleerde soorten aan boord worden gehouden, aangeland en tegen de quota worden afgeboekt. Op die manier moet teruggooi tot een minimum worden herleid. Maar wat zal er gebeuren met de populaties waarvan de vroeger teruggegooid vis goed overleefde? En hoe zal het voedselweb er uit zien als dat niet meer gebeurt? Teruggooi is namelijk al eeuwenlang een onderdeel van het dagelijkse leven op zee, en het ecosysteem heeft zich daarnaar aangepast. Toch weten we nog verrassend weinig over wat teruggooi betekent voor de vispopulaties, voor zeevogels en voor het bodemleven in zee, zowel op het vlak van bijkomende sterfte als van extra voedsel.

Teruggooi “meten” is geen sinecure

Een goed beheer van teruggooi vereist nauwkeurige data die duidelijk weerspiegelen welke factoren een invloed hebben op de omvang en samenstelling. De visgronden, de vissersvloten en hun type vistuig, de periode van het vissen en de doelsoorten van de visserij zijn de meest voor de hand liggende factoren. Toch is het inschatten van de variabiliteit van de teruggooi absoluut niet voor de hand liggend. Het registreren van teruggooi gebeurt namelijk door waarnemers die meegaan met commerciële vissers om hun vangsten te bemonsteren. Deze waarnemingen zijn duur en logistiek veeleisend, waardoor maar een beperkt aantal zeereizen bemonsterd kan worden, wat het beheer danig bemoeilijkt. De teruggooi werd onderzocht voor vier belangrijke commerciële vissoorten in de Belgische boomkorvisserij, meer bepaald voor tong, schol, kabeljauw en wijting. De voornaamste, zo niet enige oorzaak voor de teruggooi van tong en schol was hun lengte, terwijl van wijting en kabeljauw ook marktwaardige vis werd teruggegooid. De lengteverdeling van de vissen, zoals die tegenwoordig wordt bepaald door zeegaande waarnemers, blijkt dus onvoldoende om teruggooi in te schatten, en andere mogelijkheden om teruggooi te bemonsteren moeten overwogen worden. Vissers kunnen zelf hun vangsten bemonsteren. Zij zijn immers zelf altijd op zee en kunnen zo een groot aantal reizen bemonsteren. Uiteraard blijft hun commerciële activiteit primeren, waardoor het aantal soorten dat bemonsterd zal worden, beperkt is. Camera's die als een alzien oog fungeren leveren ook informatie op van vele zeereizen, maar de analyse achteraf is niet vanzelfsprekend. Ondanks intens onderzoek blijft het in kaart brengen van de teruggooi een niet te onderschatten opdracht.

Overleving van teruggegooiden organismen is een heet hangijzer

Overleven vissen vangst en teruggooi? Het antwoord op die vraag is van groot belang, omdat vissen die goed overleven niet onder de aanlandingsverplichting vallen. Dit betekent dat kleine vis toch teruggegooid zal mogen worden en dat kan een groot verschil uitmaken voor de visserij in de praktijk. Daarom werd via experimenten de overleving van tong, schol, kabeljauw, wijting, bolk en rog onderzocht na vangst met een boomkor van 4 meter. De overleving van tong was 14% na 91u observatie, terwijl de voornaamste teruggooi-soort, schol, tot 48% overleefde na 77u. Van kabeljauw overleefde 66% (88u) en van roggesoorten 72% (80u). Wijting en bolken overleefden het vangstproces niet, terwijl meer dan drie vierde van de ongewervelde bodemdieren wel. Deze studie was kleinschalig en gericht op een specifiek vistuig, maar de methode bleek té arbeidsintensief om op grote schaal te gaan toepassen. Daarom werd een techniek ontwikkeld om aan de hand van verwondingen en het testen van reflexen een inschatting te maken van hun overlevingskansen. Deze methode wordt momenteel verder onderzocht door verschillende Europese landen.

Teruggooi en het voedselweb

De studie van de overleving van teruggewoide vis hield geen rekening met 'lange-termijn' overlevingskansen. Teruggewoide organismen werden namelijk bestudeerd in aquaria en waren beschermd van natuurlijk vijanden, zoals meeuwen en Jan-van-Genten, die doorgaans in 'wolken' achter vissersvaartuigen hangen om de teruggewoide vis op te eten. Dieren die direct na teruggooi worden verorberd door meeuwen kunnen niet bijdragen tot het visbestand of dienen als voedsel voor aaseters op de zeebodem. Daarom werd een methode ontwikkeld om de relatie tussen de verspreiding van teruggooi, zeevogels en hun aantrekking tot vissersvaartuigen te onderzoeken. De methode werd toegepast op de Franse zeevisserij in de Golf van Gascogne waarbij een inschatting werd gemaakt van de hoeveelheid en de samenstelling van de teruggewoide organismen die door vogels worden geconsumeerd en dus niet meer ter beschikking komen van organismen onder water. Uit die studie bleek dat zeevogels tot één vierde van de teruggooi oppikten, en vooral dan de voedselrijke rondvis zoals wijting en horsmakreel. Hoewel de exacte rol van teruggooi doorheen het voedselweb nog niet glashelder kan worden aangetoond, is een algemene invloed van teruggooi op het ecosysteem niet te ontkennen. Denk maar aan de lage overlevingskansen van wijting, de lokale invloed van teruggooi op de bodemgemeenschap en het grote belang van teruggooi voor zeevogels.

Op naar minder teruggooi?

Het vermijden of verminderen van teruggooi in de Belgische visserij werd tot nu toe vooral onderzocht vanuit het standpunt van het vistuig. Verschillende netaanpassingen werden onderzocht met het specifieke doel om teruggooi van rondvis en ongewervelde bodemdieren te vermijden. Afgezien van het gebruik van twee aanpassingen, namelijk grote mazen in de rug van het net en een grofmazige extensie in de buik van het net, zijn deze mogelijkheden onderbenut in de commerciële visserij. Binnen de hervorming van het visserijbeleid wordt echter een vermindering van teruggooi verwacht door netaanpassingen, terwijl de efficiëntie ervan enkel op experimentele schaal is vastgesteld. Het zou dus kunnen dat de toepassing van deze methodes op vlootniveau niet het gewenste effect behalen. Daarom zullen visserijtechnische aanpassingen nauwgezet opgevolgd moeten worden, al dan niet aangevuld met bijkomende maatregelen zoals het tijds- en plaatsafhankelijk beheer van de visserij. Deze vorm van beheer steunt op het principe dat ongewenste vangst moet vermeden worden door niet te gaan vissen op visgronden waar hoge teruggooi verwacht wordt. Om dat mogelijk te maken is er een *real-time* interactie nodig waarbij vissers vangstgegevens aan elkaar doorgeven en samenwerken om te kunnen voldoen aan de aanlandingsverplichting. Ongeacht deze maatregelen is het volledig verdwijnen van teruggooi heel onwaarschijnlijk, en wat het begrijpen van de rol van teruggooi alleen maar meer nodig maakt.

List of abbreviations

- AIC: akaike information criterion
- ARS: area-restricted search
- BELSPO: Belgian science policy office
- Benthis: Benthic Ecosystem Fisheries Impact Studies
- BoB: Bay of Biscay
- BPNS: Belgian part of the North Sea
- BRP: benthos release panel
- CCTV: closed-circuit television
- CDI: catch damage index
- CFP: common fisheries policy
- CI: confidence or credible interval
- CV: coefficient of variation
- DR: Discard rate or the ratio of discards and catch
= discards/ (landings + discards)
- DQI: index of discard coverage
- EAFM: ecosystem approach to fisheries management
- EDC: experimental discard consumption
- EC: European Commission
- EU: European Union
- FAO: food and agriculture organisation
- FDF: fully-documented fisheries
- FDR: false discovery rate
- GA(M)M: generalized Additive (Mixed) Model
- GES: Good Environmental Status
- GL(M)M: generalized Linear (Mixed) Model
- ICES: International Council of the Exploration of the Sea
- ILVO: institute for agricultural and fisheries research
- INBO: research institute for nature and forest
- KM: Kaplan-Meier
- LEK: local (fishermen's) ecological knowledge
- LFD: length-frequency distributions
- LPT: landings-per-trip
- MSFD: Marine Strategy Framework Directive
- MLS: minimum landing size
- MSY: maximum sustainable yield
- MUMM: management unit of the North Sea mathematical models
- QAIC: quasi-Akaike Information Criterion
- RAMP: reflex action mortality predictor
- REM: remote electronic monitoring
- REML: restricted maximum likelihood
- RV: research vessel
- SMM: survival mixture model
- SST: sea surface temperature
- STECF: Scientific Technical and Economic Committee for Fisheries
- TAC: total allowable catch
- TL: total length
- TTM: Time-to-mortality
- VLIZ: Flanders marine institute
- VMS: vessel monitoring system
- WAKO-II: An integrated impact assessment of trammel net and beam trawl fisheries
- WGECO: working group on ecosystem effects of fisheries
- WGFTFB: working group on fishing technology and fish behaviour
- WKMEDS: workshop on methods for estimating discard survival

Prologue

Mensen,

Vergeet uw huis. Vergeet het speelgoed dat kapotging in uw kinderhanden en de films die u herbekeek. Vergeet de regering van uw land en alle hoofdsteden die u memoriseerde. Laat uw boodschappen achter in de supermarkt, denk niet aan uw falende keukenapparaten, verscheur uw groepsverzekering. Neem een trein. Vergeet uw bagage. Stap uit. Vergeet de trein. Wandel. Vergeet uw taal. Richt uw blik op de horizon. Wandel naar zee. Wandel de zee in. Ga onder.

Herinner u het water, hoe het de achtergrond vormt waartegen uw leven kan worden verteld. Hoe u met uw malse babyvoetje op een golf sloeg, hoe uw luchtbed door wilde rivieren werd vervoerd, hoe u platte keitjes over stilstaande meren liet huppelen. Herinner u dat u er op elke reis naar zocht. Dat u de zee nog wilde zien voor u ging slapen, voor u naar de luchthaven reed. Herinner u de kussen onder water. Herinner u hoe uw hart opsprong wanneer u achter het gebladerte, achter het duin, een waterval ontdekte. Het water is alles wat u dient te onthouden. U bestaat eruit, ontstond eruit en wilt er telkens naar terugkeren. U bent het niet vergeten. Ga onder.

Uit 'Vissen redden' van Annelies Verbeke

Preface

Singing from the same hymnbook

The aim and objectives in this thesis originate from two basic premises, an ethical and a historical perspective. These premises are the foundations from which several viewpoints are taken on marine commercial fisheries, and more specifically on one particular practice: discarding or returning unwanted catches of living organisms to the sea. Both perspectives, the ethical and historical, are addressed in this chapter.

Ethical perspective

'We eat wild-captured marine fish'

The starting point of this thesis is based in *shallow ecology*, being an environmental ethic discipline focusing only on the utility or usefulness of the natural environment to humans. This anthropocentric viewpoint is opposed to *deep ecology* which follows the principle that all organisms and entities in the ecosphere are part of the interrelated whole and equal in intrinsic value and the right to exist (Jacob, 1994; Leopold, 1949; Naess, 1973; Spash, 2013; Spash and Aslaksen, 2015). Deep ecology is environment-centred and based on the assumption that human activities are detrimental to nature and should be controlled ('conservationist' view). The viewpoint of this thesis, in contrast, is human-centred and based on the assumption that human activities are not necessarily or inherently detrimental to nature.

As secondary thought to the use of nature, including the seas, is related to the concept of 'sustainability' (Caddy, 1999; Frid et al., 2006; Garcia *et al.*, 2003; Pauly *et al.*, 2002; Pikitch *et al.*, 2004). Concern was raised in the 1970s on the capacity of nature to sustain human activities (Meadows *et al.*, 1972; 1992). If the human needs for food are based on the consumption of developed countries and populations continue to grow, the limits of the carrying capacity of the earth will be reached. Carrying capacity can be reached by a smooth and incremental growth rate, but the human race may also overshoot its limit drastically and ultimately decrease the overall

carrying capacity by unintentionally reducing the non-renewable resources for instance (Meadows *et al.*, 1992). The notion that a cyclical production system may be at stake raised the concern for the current generation, but also for future (human) generations. The inability to sustain ecosystem services and food production cannot be at stake, hence the concept of ‘sustainability’. The notion of ‘limits to growth’ from the 1972 report of the ‘Club of Rome’ was further put to practice through the Brundtland-report in 1987 (WCED, 1987), and the World Summit on Sustainable Development in 2002 (UN, 2002).

The concept of sustainable use of marine, renewable resources applies to fisheries and may be formulated¹ as

‘Planning, developing and managing fisheries in such a way that they meet the present needs without compromising the ability of future generations to meet their own needs.’

This notion (hereafter called ‘sustainable use’) is the basic premise to this thesis. It applies to all humans now and in the future, and is not extended to animals, implying that animal welfare, through some form of pain, suffering, enjoyment of well-being, is not acknowledged. It also implies that fishing inevitably affects the marine ecosystem, but that this intervention is accepted to the advantage of human consumption of marine resources.

The human needs are centred on food production and the ecological structures that provide them (Hilborn & Hilborn, 2012; McClanahan *et al.*, 2015). ‘Other’ ecosystem services are acknowledged as long as they do not imply a complete exclusion of human activities for harvesting marine resources. As such, the conservation of vulnerable species may be included, which relates to the intrinsic value and non-utilitarian existence of organisms. ‘Complete conservation’ is thus considered subordinate to ‘sustainable use’, which does not imply that a trade-off between conservation and sustainable use cannot be obtained. This premise is currently common practice (Jennings *et al.*, 2014) and therefore the basic premise of this thesis. This does not imply that it should not be debated.

¹ The definition of sustainable development was applied to fisheries (FAO, 1995; Garcia, 2000) and based on the concept as defined by the World Commission on Environment and Development (WCED, 1987).

Historical perspective

'We live now, and look ahead bearing memories of the past in mind'

Is 'sustainable fisheries' an oxymoron? In other words, is it not possible to predict and harvest the surplus production of fish populations year after year? Stock assessments calculate the number of individuals of a population over time. The surplus production is the net increase in the biomass from year to year. If catch equals the surplus production, then a population is harvested sustainably, as its population size remains constant. The sustainable yield at any population size is the average surplus production at that population size (Hilborn & Hilborn, 2012). The sustainable yield is at its maximum (MSY) when the population is at half of its carrying capacity under the assumption of a logistic growth curve.

The MSY-concept has been challenged in the late 1970s as summarized by this epitaph (Larkin, 1977):

Here lies the concept, MSY.

It advocated yields too high.

And didn't spell out how to slice the pie.

We bury it with the best of wishes,

Especially on behalf of fishes.

We don't know yet what will take its place,

But hope it's as good for the human race.

Pauly (1995) also highlighted that the MSY-concept was a good method to develop targets for fisheries management, but criticized its effectiveness as ecological and evolutionary considerations are factored out. He argued that the target abundance of the population which is to be sustainably harvested changes over time, and that fisheries scientists and managers link the target abundance to a reference condition which is linked to the start of their experiences rather than the pristine state of the system (the 'shifting baseline syndrome'). The baseline of ecosystem assessments is thus shifting. The true ecological cost of fisheries is thus being under-estimated when historical fishing is not accounted for. Clearly, the notion is acknowledged, also in European waters, that marine ecosystems were historically in a state with higher abundances and different biodiversity, less truncation of the age structures, less reliance on new recruits and hence a higher stability (Gu enette & Gascuel, 2012).

The perception that fisheries management should aim at the return of marine ecosystems to its pristine state, however, is unrealistic, given the existence of alternate equilibria (hysteresis). The dynamics and constant evolution of mankind as well as its environments hampers a return to the unexploited state of the marine ecosystems. However, it is acknowledged that several decades of intense fishing pressure have impaired the current state of several European waters (Roberts, 2007). The levels of decrease in stock biomass were especially affected after the industrial times. Mackinson *et al.* (2001) estimated the biomass of cod in the North Sea at 1.7 million tonnes at the end of the 19th century, whereas the biomass dropped to an approximate maximum of 280,000 tonnes in the 20th century, which was obtained after the gadoid outburst in the late sixties. Historic low levels were down to a little over 20,000 tonnes in 2006 and back to approximately 55,000 tonnes in 2012 (Cook, 1998; Horwood *et al.*, 2006; ICES, 2013a). Overall, the fish biomass and landings per unit of fishing effort decreased drastically after the industrial times (Lescrauwaet, 2013; Mackinson, 2001; Thurstan *et al.*, 2010). The biomass of large fishes weighing 4-16 kg and 16-66 kg was simulated in the North Sea through macro-ecological theory (Jennings & Blanchard, 2004). The current biomass was estimated to be 97.4% and 99.2% lower than in the absence of fisheries exploitation. Statements will not be put forward on the magnitude of the reference levels to which current management objectives should be set (historical/pristine, MSY-levels at the start of the stock assessment of a particular species for instance, or somewhere in between). Instead, the concern is merely raised that the issues of shifting baselines are hardly addressed in current management practices and that this leads to alternate states of which its stability and hence production for future generations may be at stake (Ludwig *et al.*, 1993; Pauly *et al.*, 2005; Pauly *et al.*, 2003; Pauly *et al.*, 2002; Pitcher, 2001; Pitcher & Pauly, 1998).

The thesis does not intend to investigate the issue of shifting baselines. In contrast, it takes the current fishing practices and interactions with the current state of the marine ecosystem as starting point. Advances to sustainable use of marine resources are exclusively related to current management objectives, such as fisheries' discards. Discards are the return of unwanted organisms to the sea and are considered a wasteful practice. The thesis is focusing very much on the provision of building blocks to account for the consequences of discarding in an ecosystem perspective and expects by doing so, that this increased understanding will increase our ability to use marine resources sustainably, together with insights on exploitation patterns (Law *et al.*, 2015). The investigations of this thesis will thus not allow constructing and engineering pre-exploitation levels of fish community structure and function, but may help to look at its current state and prevent it from deteriorating further. The second premise focused on maintaining what we have and gradually moves towards 'sustainable use' rather than rebuilding stocks to levels from the past.

1 Introduction

Every man takes the limits of his own field of vision for the limits of the world. - A. Schopenhauer

1.1 Fishing effects on the marine ecosystem

1.1.1 *Historical recognition of the ecosystem effects of fishing*

Sustainable use of marine resources will always result in some level of impact, in the least the extraction of the fish itself. Since the early history of fishing, men have been concerned with this awareness. In 1376 a petition was presented to the British Parliament to call for the prohibition of the 'wondyrchoum'. This fishing gear was the earlier version of the beam trawl, being a net with a wooden beam to keep it open. The concern was raised that it caught too many small fish and would damage the UK fisheries. Towed gears were also prohibited in Flanders in 1499, and similar legislation followed in the Netherlands and in France, because the gear was thought to destroy structures that serve as hiding places for fish (Rabaut, 2009).

The current application of the beam trawl fisheries is based in the 19th century when the gear was first applied from sailing vessels. The application of the gear was facilitated from 1881 onwards when it was first deployed from steam trawlers, which were able to tow a heavier beam trawl with tickler chains (Mackinson, 2001). Fishing vessels were largely replaced by motor trawlers around the 1930s, towing primarily otter trawls. Beam trawling for flatfish was re-introduced after the Second World War, and became significant from the sixties onwards (Philippart, 1998).

The first investigations on the ecological impact of fishing were initiated by fishermen, worried that the gear may harm seabed habitats and impair future fishing opportunities (Jones, 1992). Graham (1955), for instance, investigated the trawling effects on macrobenthos by comparing the catches in a fished and a non-fished area. The author concluded that fishing did not affect benthic habitats. Later on in the 1950s to the 1970s, research efforts were mainly driven by investigations to increase the catch efficiencies of the gear (including selectivity), rather than the study of its adverse effects on the seabed. However, the use of more and heavier tickler chains, the higher overall weight of the gear (from 3.5 tonnes in the late 1960 up to 10 tonnes in the early 1980s) and the increase of the fishing fleet stimulated debate on the ecological effects of fishing. In the seventies and eighties, several studies investigated the effects of bottom trawling on the seabed (de Groot, 1984; de Groot &

Apeldoorn, 1971), as well as the potential of discards to survive the catching process (de Veen *et al.*, 1975), although studies on fuel efficiency and species and size selectivity dominated research efforts in these years (Walsh *et al.*, 2002). It was not until the early 1990s that impact studies gained full momentum, indicated for instance by the European project 'IMPACT-I' and 'IMPACT-II' (de Groot & Lindeboom, 1994; Lindeboom & de Groot, 1998). These studies investigated a wide range of bottom trawling effects: catch efficiency, catch composition, survival of the discards, mortality in the tow path, scavenging behaviour after a trawl had passed, and so on. They still form the basis for many different impact studies.

1.1.2 Two main methodologies to study the ecosystem effects of fishing

Ecosystem effects of fishing may be subdivided into short-term, direct effects versus long-term effects. Direct effects are typically easier to study as they can be practically investigated during experimental trials. They are related to a particular fishing metier, as characterised by gear, target species, place and time. The relevance of these experiments to the fleet or ecosystem level requires an in-depth and detailed understanding of the propagation of the short-term effects through time and space in order to know whether the short-term, direct effects have a substantial impact on the ecosystem in the longer run. A classic example is the study of short-term survival of discarded fish. The ability of a discarded plaice to survive the catching process may be 'easily' studied by holding the organisms in on-board observer tanks. The interpretation of the result on short-term survival for the long-term viability of the plaice stock is much more difficult to predict, because of the rapidly increasing number of variables that come into play. The study of short-term experiments nevertheless aids to identify how fishing pressures vary and hence, how they may be mitigated. Their efficacy is what is to be monitored 'posthoc'. Long-term trends in ecosystem structure and function are, in contrast, what really matters, but the causal links between the evolution of ecosystems and the natural and/or human-induced disturbances are highly complex and difficult to relate to one another (Kaiser, 1998). Therefore, a combination of short-term and long-term studies may complement one another and helps TO understand the ecosystem and fishing pressures upon it. Both, short-term and long-term effects of fishing on the ecosystem are studied in great detail since the 1990s (Clark & Frid, 2001; Frid *et al.*, 2000; Greenstreet & Hall, 1996; Heessen & Daan, 1996; Witbaard & Klein, 1994).

This thesis takes the approach of investigating short-term, direct effects and the causes of its variability, to enable policy decision support. The link to long-term effects was not studied, but should be kept in mind to evaluate the relevance of short-term effects to the overall ecosystem.

1.1.3 Different types of ecosystem effects of fishing

The short-term, direct ecosystem effects are tightly linked to the fishing gear (gear drives the intensity of interactions of fishing with the ecosystem) and vary by the way that a gear is deployed (frequency, time and location of ecological effects).

Demersal gear, for instance, directly affects the physical and geochemical characteristics of seabed habitats as well as its associated benthic communities. The physical and chemical impacts of bottom trawling on the seabed range from smothering, resuspension of sediment and nutrients, compaction, to abrasion and changes in oxygenation (Depestele *et al.*, 2015; Løkkeborg, 2005; Smith *et al.*, 2003; van Denderen *et al.*, 2014). The biological impacts are related to altered production and biodiversity as a result of the induced mortalities, as well as reduced fitness, changes in food availability, and so on. (Hiddink *et al.*, 2006; Hiddink *et al.*, 2007; Hughes *et al.*, 2014; Lambert *et al.*, 2014; Shephard *et al.*, 2010; Shephard *et al.*, 2014; Shephard *et al.*, 2011; Tillin *et al.*, 2006). The state-of-the-art physical and biological impacts of bottom trawling on benthic habitats are reviewed in several publications (de Groot & Lindeboom, 1994; ICES, 2007b; Jennings & Kaiser, 1998; Kaiser *et al.*, 2006; Lindeboom & de Groot, 1998; Løkkeborg, 2005; Polet & Depestele, 2010). Although greatly studied, the effects on benthic habitats continue to be a topic of great debate, which ranges from recommendations to ban bottom trawling to the opposite, namely maintenances of fishing practices as it creates food for the harvested species (ICES, 2007a; van Denderen *et al.*, 2013).

Direct, short-term effects other than the physical, geochemical and biological impacts on benthic habitats may include effects related to noise and visual disturbance of the fishing vessels and gear (Deerenberg *et al.*, 2010; Schakner & Blumstein, 2013), as well as CO₂-combustion and litter production (Barnes *et al.*, 2009; Galgani *et al.*, 2000).

The most obvious effects of fishing are related obviously to the catch. Catching species that fishermen bring to the market (marketable species) is not considered as an adverse impact when the fishery is managed sustainably. Marketable species comprise individuals of a 'commercial' species which are above the Minimum Landings Size (>MLS). Commercial species are species which fishermen decide are valuable for them to land on the market. Marketable species may be subdivided into target and by-catch species. A target species is a species that fishermen intend to catch, i.e. their fishing behaviour is targeting this species. A target species in beam trawl fisheries is generally common sole (*Solea solea*, hereafter called sole) but may also be European plaice (*Pleuronectes platessa*, hereafter called plaice) or brown shrimps (*Crangon crangon*, hereafter called shrimp unless otherwise mentioned). Bycatch species are defined as the species that are caught next to the target species. The target species of the flatfish-direct beam trawl fleet are sole and plaice; but

by-catch species, such as brill (*Scophthalmus rhombus*), brill, common dab (*Limanda limanda*, hereafter called dab), Atlantic cod (*Gadus morhua*, hereafter called cod) and whiting (*Merlangius merlangus*) are also landed (Daan, 1997). Target behaviour may be estimated from the increased presence in locations and periods where the catch of the target is higher (Quirijns *et al.*, 2008). Bycatch species is the part of the catch that is unintended. They may be retained and landed or they may be returned to the sea.

The animals that are caught and brought on deck only to be returned to the sea are defined as 'discards'. Discards exclude any plant material. Theoretically, discards include all animals. However, in this thesis, the accidentally caught species of the bird and marine mammals group are excluded from the definition of discards. The catch of these 'charismatic' species is generally referred to as 'accidental' catch or 'by-catch'. The terminology is blurred in scientific literature.

Discards are thus defined as

'the *animal* part of the catch (excluding plants) that is *unwanted*, except for marine mammals and birds'.

Discards may include undersized marketable species, commercial species \geq MLS that are unwanted due to legislation, low market value or any other reason, and non-commercial species ranging from benthic invertebrates to cephalopods and fish. The definition of discards is broad, and is generally further specified by the objective of the study, which mainly includes specification on the fishery that causes the discarding and the animals that are compromised in discard observation (see Box: 'the conundrum of discard terminology').

In conclusion, the ecosystem effects of fishing are bewildering and range from physical, geochemical to biological effects. Most studies of short-term, direct ecological effects focus on the catch (landings and discards) of the target species and by-catch species, and the consequences for the population of these species. The catch of non-commercial species also receives research attention, and many studies evaluated also the by-catch of marine mammals and seabirds. Short-term, direct effects are also related to habitats, being it purely physical or chemical, or include biological aspects. Long-term ecological studies may primarily be related by linking fishing pressures to populations of the commercial species, followed by non-commercial species as well as their interactions in food-web models. This thesis focuses in a first instance on discards, but may open doors to link the results to mitigation options, as well as the longer-term studies of the effects of discards in the food web.

Box: 'the conundrum of discard terminology'

The discard conundrum starts when discards are being quantified. This thesis among other studies illustrate that the quantification of discards is rarely realised following the definition above. The presented discards rarely include all discards, but generally refer to *partial* discards, i.e. the discards of a particular 'group of animals' in a particular 'fishery'.

A *fishery* may be summarized in different ways: by country, by gear, by area, by season or by a mixture of these and other criteria. The discards of the Belgian fleet for instance comprises mostly beam trawl fisheries, but also other fishing gears with different catch efficiencies (otter trawls, gill netters). Also within a gear type there is variation in gear rigging and/or deployment. Beam trawls, for instance, have a codend mesh size of ≥ 80 mm for flatfish-directed, and ≥ 20 mm for shrimp-directed fisheries. Accounting for the fishery is indispensable when interpreting a discard estimate.

The categorization of the *group of animals* also varies significantly according to the objective of the study or the type of information that is available. Taxonomic categorization is a regular practice, e.g. fish versus benthic invertebrates, or the discards of a particular species within a region and period. However, the importance of discards is mostly related to the discards of one single species in a particular area, i.e. the stock. A stock comprises all the individuals in a well-defined spatial range, which are part of the same reproductive process. It is self-contained, with no emigration or immigration of individuals from or to the stock, and could be applied using genetic or phenotypic markers that are inherited (Booke, 1999; FAO, 2005). Stocks are generally referring to a population or subpopulation of a commercial species. A species is 'commercial' when it is being sold on the market.

The discards of commercial species are dynamic over time, because it depends on volatile behaviour of buyers and consumers at the market, as well as the interest of fishermen and the catchable biomass. Discarded catches may be converted to landed catch if new markets or processing techniques are developed (Harrington *et al.*, 2005). The commercial interest also varies by fishery. Atlantic herring (*Clupea harengus*) is sold on the market, but is not a target or *traditional* by-catch species in beam trawl fisheries, even though it may occasionally be caught and landed (DLV, 2014). Some species are caught as bycatch in one fishery while they constitute a major component in another fishery. These differences in commercial interests lead to different obligations for monitoring the catch. For instance, monitoring the discards of beam trawl fisheries must not include the accidental catch and discards of pelagic species (EC, 2008a, 2008b, 2009), whereas herring-directed fisheries do observe discards of Atlantic mackerel (*Scomber scombrus*, hereafter called mackerel) for instance (Borges *et al.*, 2008). The discards of commercial species in beam trawl discards may thus comprise a different suite than those of pelagic fisheries.

Moreover, the difficulty in precisely and accurately defining discards is further complicated by the methods of quantifying discard estimates. Discarded numbers, discarded weight, discarded volumes or discard rates are the most common unit. Discard rates are the ratio of the discards to the total catch (discards + landings). When discard rates are expressed in numbers, then the estimates are generally higher than in weight, because discarded individuals are generally smaller than the landed individuals.

The interpretation of discards through this thesis requires specific attention of these issues, especially because of the differences in objectives across the presented studies (see further). Interpretation needs to account for the objectives of the study, and the specific categorization of animals and fisheries as a consequence of this objective.

1.2 The interplay of science and fisheries management

Fisheries' science is at the interface between the fishing industry, fisheries managers, and other scientists among other stakeholders including non-governmental organisations. The role of scientists is to provide advice that is 'objective' to underpin policy objectives, or at least meets 'the' methodological standards of scientific requirements. Science is to provide the eyes on the marine ecosystem for policy makers to see how policy objectives can be best defined and which management measures may assist in achieving these objectives. The knowledge base of scientists may be greater than the knowledge base applied in policy (science may be ahead of policy), or the scientific knowledge base may be insufficient to implement policy objectives (science may be lagging behind to support policy) (Figure 1.1; Rice, 2011a).

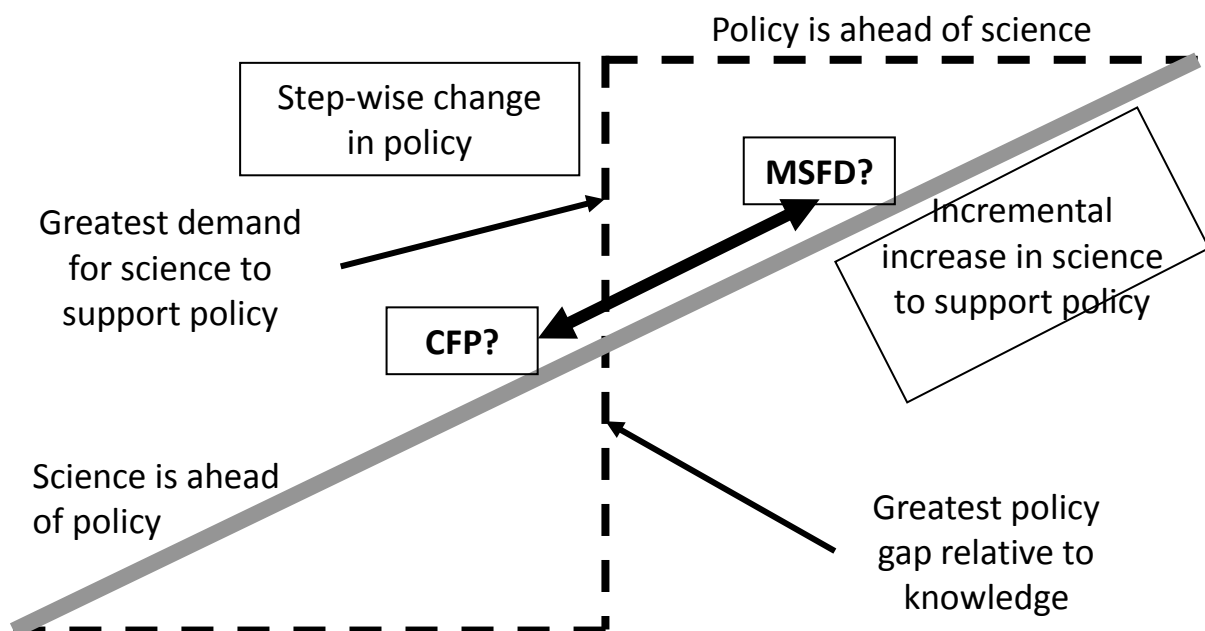


Figure 1.1 Thought exercise of the science-policy interface, modified from Rice (2011a). The knowledge base in science changes incrementally, while policy objectives are set in time steps. The potential discrepancy between the Common Fisheries Policy (CFP) and Marine Strategy Framework Directive (MSFD) illustrates in particular the differences in political advances for the European context.

European policies have currently moved towards the implementation of the EAFM by the Marine Strategy Framework Directive (MSFD, EU, 2008) and the Reform of the Common Fisheries Policy (EU, 2013a). Scientific advances indicate that the current focus on single stocks is insufficient, but does not have a 'sufficient' knowledge to implement the ecosystem interactions into a practical management framework. Science, however, aids policy to see how this may be performed in the nearby future. While the Common Fisheries Policy (CFP) is largely based in the single species framework, it recognizes the requirements to move towards an 'Ecosystem Approach', which is practically being implemented in Europe through the MSFD. Science however lags behind several

MSFD-aspects, such as the implementation of food web indicators for instance. This thesis contributes to advancing the views of scientists on seeing how discards may be affecting single species as well as the marine ecosystem. This is in the long run assisting policy support on how discards (as a component of fishing mortality) may affect the sustainable use of marine resources within an 'ecosystem perspective'.

1.3 Fisheries management and the focus on single species populations

1.3.1 *Stock status and fleet capacity management*

Fisheries management is traditionally focusing on managing single species populations. Globally the exploitation of fish populations has reached its limits in the late 1980s, and the recovery of depleted stocks has become a cornerstone of fisheries management (Pauly *et al.*, 2005; Worm *et al.*, 2006; Worm & Branch, 2013; Worm *et al.*, 2009). The current global trend is diverging between fish populations that are scientifically assessed and those that are not. Unassessed stocks are typically further declining (Costello, 2012), whereas managed fish stocks are generally improving. Rebuilding fish populations is especially a trend for larger stocks (Hilborn & Ovando, 2014). However, it is also being argued that the stock improvements typically restrict themselves to isolated cases and that some ecosystems have shifted without any recovery of the stock (Howarth *et al.*, 2013), such as happened to the cod stock off the coasts of Canada (Mullowney & Rose, 2014; O'Boyle & Sinclair, 2012). The status of the world's fish abundance is still being debated (Pauly *et al.*, 2013).

The status of fish populations in European waters have also shown mixed results (see below), but the European Union and its Member States have committed themselves to act against the continued decline of many fish stocks at the World Summit on Sustainable Development in Johannesburg in 2002 (UN, 2002). The commitment implied that The CFP would be improved by 2015 through the adaptation of exploitation rates so as to ensure that within a reasonable time-frame, the exploitation of marine biological resources restores and maintains populations of harvested stocks above levels that can produce the maximum sustainable yield. The exploitation rates should be achieved by 2015, although achieving those exploitation rates by a later date is allowed only if achieving them by 2015 would seriously jeopardise the social and economic sustainability of the fishing fleets involved (EU, 2013a). The social and economic viability is at stake in several Member States, as it has been estimated that the cost of fishing to the public budgets exceeds the total value of the catches (European Commission (EC), 2009). This implies that the exploitation rates to obtain MSY may be achieved after 2015, but in any event no later than 2020.

The highest impact of fisheries management measures in European waters took place from the late 1990s onwards up till recent years through the decrease in fishing pressure and the reductions in fishing mortality rate of assessed stocks as a result of decommissioning schemes and high fuel prices (Tidd *et al.*, 2011). However, these cuts in fishing effort have not necessarily led to a recovery of the biomass of all assessed stocks (e 1.2; Gascuel *et al.*, 2014). Appendix 11.1 illustrates how fisheries' catches evolved through time for a selection of species that are landed by Belgian fisheries.

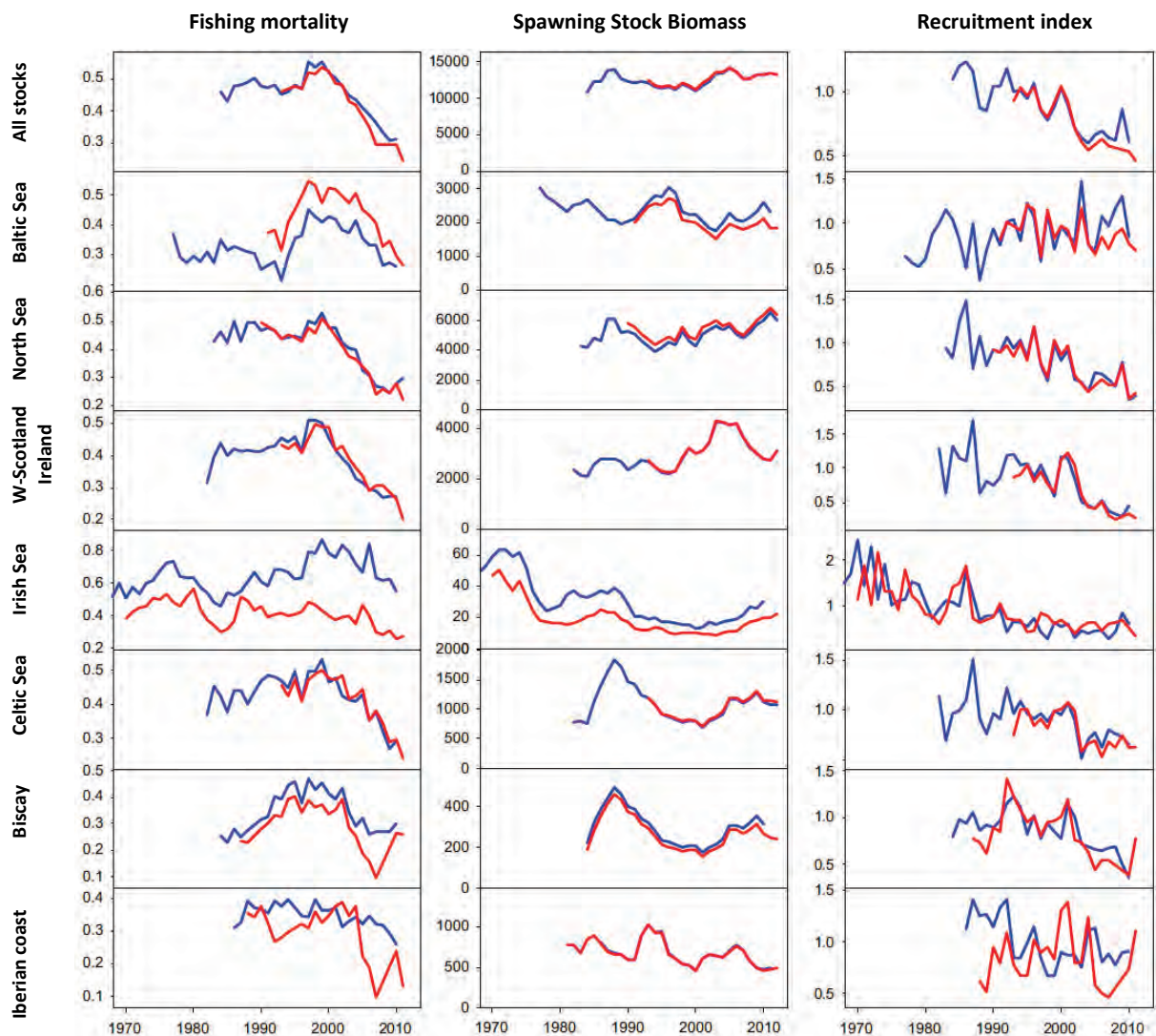


Figure 1.2 Trends in stock-based indicators: mean fishing mortality (left column, per year), total spawning stock biomass and the mean recruitment index R (R was calculated by stock as the ratio of recruitment in year y divided by the mean recruitment of that stock over 1990-2000). Indicators are based on 57 stocks assessed by ICES in European and regional seas. The red line refers to all stocks assessed in 2012, while the blue line is the longest available time series including at least 60% of assessed stocks. A strong decreasing fishing mortality has not yet led to opposite, clearly increasing trends in spawning stock biomass (After Gascuel *et al.*, 2014).

Some studies stress the lack of recovery. Froese *et al.* (2015) calculated the Good Environmental Status (GES as defined in MSFD Descriptor 3; EU, 2008) of stocks in German marine waters for fully-assessed as well as data-limited stocks. Overall, the indicator showed that only 3 out of 19 stocks

were above the limited reference points in 2011. These authors highlight that continuing efforts are needed, as well as the needs to include data-limited stocks in assessments. Another study also highlights that ten European fish stocks are 'outside safe biological limits' (Steadman *et al.*, 2014). Traditional fisheries management was criticized and the authors stressed the need to manage stocks 'tougher', as well as the fact that 54% of the European stocks are considered data-deficient. The population of sole in the Irish Sea was one of the examples. The stock is primarily targeted by the Belgian beam trawler fleet. The authors argue that the reduction in fishing pressure is insufficient, and argue that environmental gradients on the recruitment success potentially affect the stock rebuilding as well.

Other studies, in contrast, emphasize that bringing back stocks to Maximum Sustainable Yield is not something one can be achieved overnight after several decades of overexploitation, but that signs of improvement are present during the last ten years (Cardinale *et al.*, 2013). The North Sea plaice stock may be an example where the reduction of fishing pressure has resulted in increased populations (ICES, 2014a; Trenkel *et al.*, 2015).

The current status of several fish populations, including those exploited by Belgian beam trawlers, show that efforts to enhance stock status are essential, and that reduction in fishing pressure and management of fleet over-capacity have a significant but delayed effect.

1.3.2 *Strengthening discard management to improve stock status*

Fisheries discards are perceived as a waste of natural resources by public perception and policy. The FAO Code of Conduct for Responsible Fisheries (FAO, 1995) states in Article 8.4.5:

'States, with relevant groups from industry, should encourage the development and implementation of technologies and operational methods that reduce discards. The use of fishing gear and practices that lead to the discarding of catch should be discouraged and the use of fishing gear and practices that increase survival rates of escaping fish should be promoted'.

Article 8.5.1 of the FAO Code of Conduct promotes fishing gear selectivity to obtain this aim:

'States should require that fishing gear, methods and practices, to the extent practicable, are sufficiently selective so as to minimize waste, discards, catch of non-target species, both fish and non-fish species, and impacts on associated or dependent species and that the intent of related regulations is not circumvented by technical devices. In this regard, fishers should cooperate in the development of selective fishing gear and methods. States should ensure that information on new developments and requirements is made available to all fishers'.

Fisheries management in the European Union is outlined in the CFP. In the first years of this century the Green paper on the future of the CFP (European Commission (EC), 2001a) stressed that sustainability of a high number of stocks would be threatened if the current levels of exploitation were maintained. Over-fishing, discarding and fleet over-capacity were identified as main contributors of the problems in fish populations in European waters. The reduction of discards was highlighted as one of the major ways to return to healthy fish stocks:

'Adoption of stronger technical measures to protect juveniles and to reduce discards including pilot projects for measures not applied until now such as discard bans'.

A reform of the CFP was due in 2012, as every common European policy is to be revised every ten years. The Green Paper on the reform of the CFP in 2009 (European Commission (EC), 2009a) indicated that

'[...] the objectives agreed in 2002 to achieve sustainable fisheries have not been met overall.'

and that

*'[...] most fish stocks have been fished down. 88 % of Community stocks are being fished beyond MSY: this means that these fish populations could increase and generate more economic output if they were left for only a few years under less fishing pressure. 30 % of these stocks are outside safe biological limits, which means that they may not be able to replenish. **European fisheries today depend on young and small fish that mostly get caught before they can reproduce.** For instance, 93 % of the cod in the North Sea are fished before they are mature. This overall picture conceals considerable variations by marine region and species. Nonetheless, European fisheries are eroding their own ecological and economic basis.'*

The question arising from this management failure is how the reformed policy objectives can be defined regarding ecological, economic and social dimensions of sustainable fisheries? The Reform also questioned how objectives may give guidance in the short term in a clear, prioritised manner and ensures the long-term sustainability and viability of fisheries? The Reform shall focus on matching fishing opportunities with fleet (over-)capacity and continue its focus of discard reduction. Discard reduction culminated in the present day with the landing obligation, which serves as the main driver to partially eliminate discards of certain commercial species (EU, 2013a):

*'Measures are needed to reduce the current **high levels of unwanted catches** and to **gradually eliminate discards**. Unwanted catches and **discards constitute a substantial waste and negatively affect the sustainable exploitation of marine biological resources and marine ecosystems** and the financial viability of fisheries. An obligation to land all catches ('the landing obligation') of species which are subject to catch limits [...], made during fishing activities in Union waters or by Union fishing vessels should be established and gradually implemented and rules that have so far obliged fishermen to discard should be repealed.'*

The Reform of the CFP clearly expects the reduction of discards, which has up to now only partially or not been achieved for several fleets (Enever *et al.*, 2009; Heath *et al.*, 2015). Fisheries management in Europe expects that this reduction of discards will rebuild fish stocks to MSY-levels in combination with other management measures such as matching fleet capacity with fishing opportunities, increasing the accountability of stakeholders in fisheries management and regionalisation, in part linked to geographical sea areas (Figure 1.3).

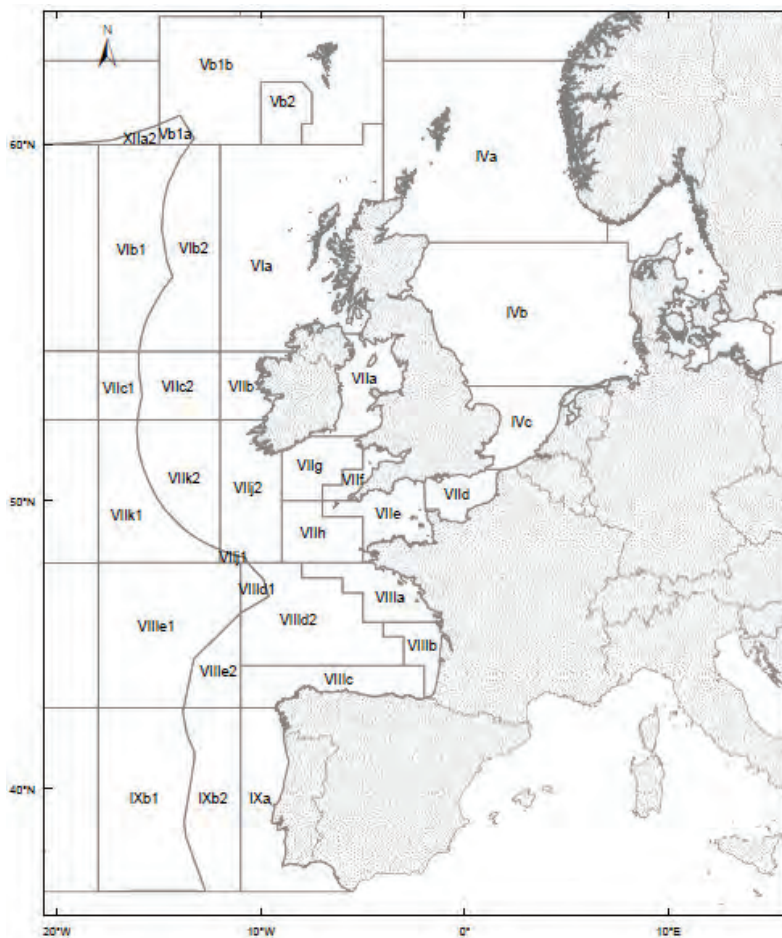


Figure 1.3 Fishing areas in the NE Atlantic as delineated and used by ICES, such as Subarea IV (North Sea) and its ICES Divisions IVc (southern North Sea), IVb (central North Sea) and IVa (northern North Sea).

1.4 Scientific understanding of single species populations within the marine ecosystem

The sustainable harvest of marine resources is predominantly managed by fishing mortality as set by TACs (Total Allowable Catches or landings quota) or by catch quotas in the reformed CFP, but sustainable, long-term exploitation also depends on unknown consequences of (1) changed population structures (age-truncated structures), (2) changes in predator-prey interactions due to altered size and species composition of populations, (3) changes in both resource and interference competition, and (4) changes resulting from disturbed habitats. These concerns may have to be addressed as well if fish populations are to recover. Habitat changes and its importance to fish stocks are complex to quantify and relate to fishing pressure. Recent studies have mapped fish habitat and indicated how important habitat loss and/or degradation may be at various spatial scales (Johnson *et al.*, 2013; Le Pape *et al.*, 2014). The relationship between fishing pressure and disrupted population and community structure is clearer and was addressed below in more detail.

1.4.1 Disrupted population structures decrease stability through time

The stock status in one year is but one of the primary targets for sustainable use of marine resources. Sustainable exploitation also requires long-term stability of the stock status. Removal of biomass reduces the population biomass and affects biomass production. Long-term sustainable exploitation requires that fishing effort is such that the population biomass and biomass production is stable over the years, but this is rarely the case. Fishing causes a higher temporal variability in populations than can be found in time series of the abundances of unexploited populations (Hsieh *et al.*, 2006; Figure 1.4). The cause of this variability was not related to the yearly variability in fishing pressure or to the fishing-induced decreases in mean body size and/or age, but to the age-truncation of exploited populations caused by selective fishing (Anderson *et al.*, 2008; Rochet & Benoît, 2011). Age-selective fishing increased the relative demographic contribution of recruits and amplifies the destabilizing effect of environmental variability, as demographic parameters are coupled with environmental variability (Sugihara *et al.*, 2011). The stock fluctuation in the Irish Sea sole stock may illustrate the high temporal variability of the sole biomass, at least if the sole population recovers. Fishing causes non-linear dynamics in the marine ecosystem, which makes short-term prediction highly complex (Glaser *et al.*, 2013). The size-selective extraction of fish from the ecosystem complicates management of single species populations and increases the risk of tipping points at which a sudden shift to a contrasting dynamic regime may occur (Scheffer *et al.*, 2009). The temporal variability in size-selective extraction makes it also more difficult to match the demand and supply at fish markets and to keep fish prices constant through time (Pinnegar *et al.*, 2002; Tsikliras & Polymeros, 2014).

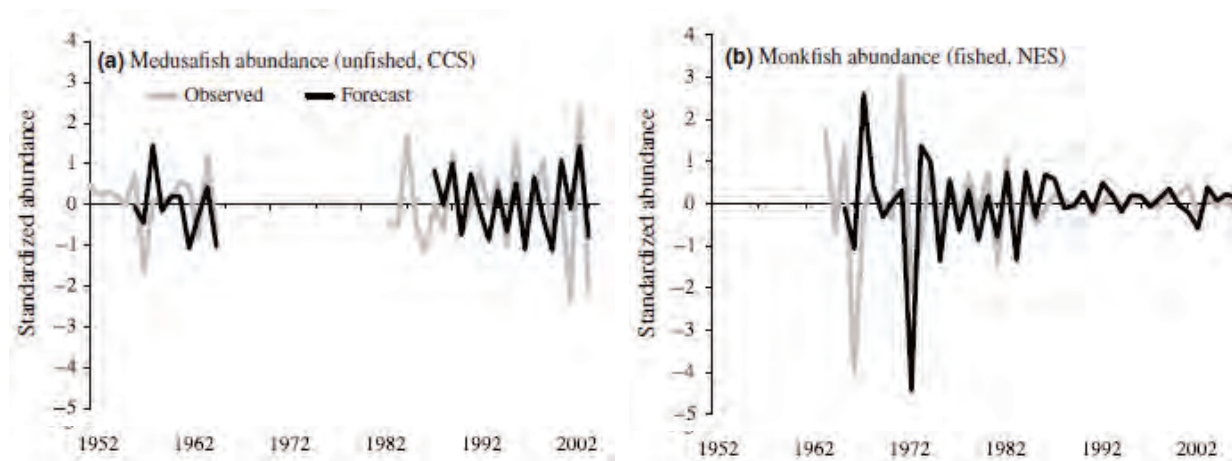


Figure 1.4 Illustration of the variability in (observed and forecasted) standardised abundance of an unfished species in the Californian Current System (CCS) (left) and a fished species in the Northeast U.S. Continental Shelf System (NES) (right). The unfished species followed linear dynamics, but evidence of non-linear dynamics was collected to forecast the abundance of the fished species. Non-linear dynamics imply that small changes in input variables (e.g. fishing mortality) may lead large changes in response variables such as yield. The temporal variability in abundance is higher for the fished species. Further details can be found in Glaser *et al.* (2013).

Current fisheries management regimes are based on the concept of growth or recruitment overfishing, i.e. the loss of yield when small fish are caught. The concept of avoiding juvenile fish was introduced in the 1950s (Beverton & Holt, 1957) and was implemented in fisheries management by a simple rule (Myers & Mertz, 1998): the ‘spawn-at-least-once’ policy. This policy assumes that fish stocks will not collapse if fish can reproduce at least once before they are being harvested. The selective removal of large fish causes age-truncation of populations and reduces the proportional abundance of larger individuals which have an exponentially higher fecundity and which produce larvae with substantially better survival than smaller fish (Berkeley *et al.*, 2004; Birkeland & Dayton, 2005).

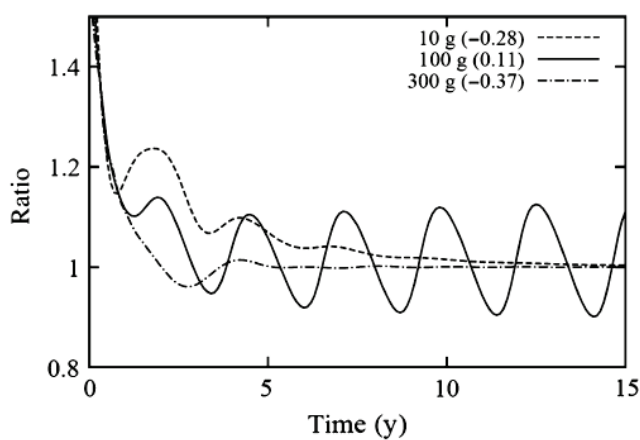


Figure 1.5 Destabilisation caused by size-selective fishing. A simple dynamic model of size spectra illustrated that the size at which fish can be caught (size-at-entry) affects steady states of the stock biomass. The ratio on the Y-axis is the stock biomass at time t divided by the steady state stock biomass. The lines show the times series of the ratio, i.e. the deviation from steady state, following a fishing event. The stock did not stabilise within 15 years when size-at-entry was close to the size at maturation (150 g), as is the case in the ‘spawn-at-least-once’ policy (continuous line), but deviation from steady state were substantially decreasing in the balanced harvesting scenarios (dashed lines). Further details can be found in Law *et al.* (2015). *al.*, 2015).

Reducing the boom and busts in exploited fish populations through changes in the size-selectivity paradigm has recently gained attention. In contrast to the ‘spawn-at-least-once’ policy and its subsequent minimum size regulations, the concept of balanced harvesting emerged, whereby fishing mortality is set proportional to productivity (Garcia *et al.*, 2012; Garcia *et al.*, 2015; Jacobsen *et al.*, 2013; Law *et al.*, 2014). Balanced harvesting helps retaining the approximate power-law size-structure² of natural systems and is therefore less likely to destabilize steady states (Law *et al.*, 2015; Figure 1.5). Consensus has not been reached within the scientific community about the theoretical rationale (Froese *et al.*, 2015) or its implementation (Burgess *et al.*, 2015).

² The power-law size spectrum in unexploited marine systems implies that the density of organisms of a given mass (per unit volume per unit body mass, $\phi(w)$) follows a power-law function of body mass w with an exponent γ close to -2 and a constant β : $\phi(w) = \beta w^\gamma$ (Cohen *et al.*, 2012; Datta *et al.*, 2011; Marquet *et al.*, 2005).

1.4.2 *Disrupted species interactions change the potential for population recovery*

The 'spawn-at-least-once' policy imposed size-selective fishing as an important goal in the sustainable exploitation of fisheries. Age-truncation disrupts the population structure and may compromise its potential for recovery. Fishing is also species-selective. The selective removal of species and/or functional groups (piscivore, benthivorous guilds, etc.) is another factor that influences the stability of species populations, communities and marine ecosystems (Bell *et al.*, 2014) by altering interactions between species (predator-prey, competition).

The effects of mortality of a particular species are thus not only important for the fish population experiencing the mortality. The reduced abundance of that species also implies that its ecological function is reduced, unless it is being replaced by another species. When this is not the case, the effects of its reduced abundance and functioning may affect other populations, and potentially propagate throughout several organisational levels of the ecosystem. As such, the interactions between fish species may hamper the sustainable exploitation of one stock versus another one, as illustrated by the cod stock in the Baltic Sea (Eero *et al.*, 2012).

The cod stock in the entire eastern Baltic Sea is half of its stock size observed in the early 1980s. However, the density of cod in the south-western part of the Baltic Sea is close to the highest level since the 1970s, while recovery is low in the northern part of the Baltic Sea. The higher abundance of cod in the south-western Baltic Sea does not lead to a higher, overall productivity in the Baltic Sea, because of the low weight and condition of the adult cod. The poor condition of the adult stock in the south-eastern Baltic Sea was related to food limitations. The main prey of cod is European sprat (*Sprattus sprattus*, hereafter called sprat) and herring. These two pelagic species are also exploited by fisheries and are in contrast to cod, more abundant in the northern areas and less abundant in the south. Sprat and herring suffer increased predation mortalities from cod in the south, as well as fishing pressure. Sprat and herring are at low levels in the south-western Baltic Sea, and hamper the recovery of the cod stock in the entire Baltic Sea. This example shows that the recovery of cod may take place when fishing pressure on herring and sprat is relaxed or displaced to the northern Baltic Sea. The prey-predator interactions of cod with other species and the changes in structure of the Baltic ecosystem have shown to be important in the recovery of cod. The Baltic Sea example illustrates how the recovery of a fish population may be limited by other species, but did not necessarily lead to an alternate ecosystem structure.

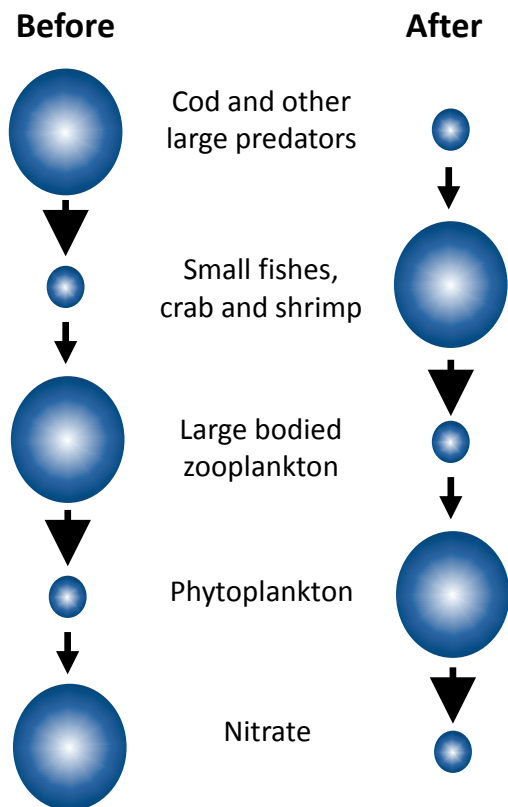


Figure 1.6 Overfishing of cod and other large predatory populations released small fish and invertebrates from predation pressure, which in turn led to a decline in large-bodied zooplankton species (the preferred food of shrimp and crabs) and further cascading effects on the food web (after Scheffer *et al.*, 2005).

The formerly cod-dominated ecosystem off the Canadian east coast shows that ecosystems with reduced resilience may also switch to a contrasting state with cascading effects on all trophic levels as a consequence of strongly reduced populations of top predators (Frank *et al.*, 2005; Scheffer *et al.*, 2001; Scheffer *et al.*, 2005). Several local cod stocks off the Canadian east coast have collapsed without recovery. The mortality caused by fisheries affected the survival and productivity of the cod population, which is a species from the upper trophic levels of the food chain or web. The removal completely restructured the food web and is manifested by inversed changes in biomass between adjacent pairs of trophic levels (Figure 1.6). The ecosystem changed into a system dominated by planktivorous, forage fish and macro-invertebrates (Frank *et al.*, 2005; 2011).

The alternate ecosystem state led to an increase in many other low-trophic level fisheries, such as Northern shrimp (*Pandalus borealis*), sea urchins (*Strongylocentrotus droebachiensis*), sea cucumbers (*Cucumaria frondosa*), several species of crabs, as well as fisheries on typical scavenging fish species such as hagfish (family Myxinidae) (Andersen *et al.*, 2008; Ellis *et al.*, 2015). Low-trophic levels such as invertebrate species also influence ecosystem structure and function significantly. The direction and magnitude of the responses depend on their function in the system and the exploitation rate (Eddy *et al.*, 2015), but it is clear that concerns on the stability of their exploitation are also required.

While fisheries management and fisheries' science is primarily directed towards the populations of several single species population, typically with a commercial value, the marine scientific community at large also undertakes efforts to understand broad-scale patterns resulting from changes in interactions within and between species populations. These patterns may be captured by general concepts such as the productivity, stability and biodiversity of populations and communities, and the resulting changes from fishing and other drivers such as climate change.

The relationships between productivity, stability and biodiversity are bi-modal and may be influenced in both directions (Worm & Duffy, 2003). The risk of stock collapse, its potential of recovery and its stability through time has been shown to decrease exponentially with declining diversity in coastal as well as large marine ecosystem. It was suggested that larger numbers of species are needed to reduce the temporal variability in ecosystem processes in changing environments (Worm *et al.*, 2006). The effect of biodiversity on production is, however, controversial and initiated considerable debate (Fridley, 2001; Murawski *et al.*, 2007). A recent study has put the Biodiversity-Ecosystem Functioning relations into a different perspective. The study comprised a complex model with >1000 species. Production, expressed as fisheries' yield, reached its highest levels when about 60% of the fish species are extirpated (Fung *et al.*, 2015), implying that (1) biodiversity is important, but also that (2) productivity is at its maximum at lower levels of biodiversity. The ongoing debate between biodiversity and ecosystem functioning, such as production, illustrates in the first place that different ecosystem services require trade-offs. Society has to decide whether it aims at a maximum biodiversity or a maximum production, or whether a trade-off between both objectives can be found through spatial management or other management mechanisms. Of relevance to this thesis is the clear influence of species interactions on the biomass as well as production of commercial fish stocks. The influence of biodiversity on both production and stability illustrates that rebuilding a single species population cannot be achieved by the sum of individual single species rebuilding plans.

The difficulty in rebuilding an entire fish community was illustrated by a modelling exercise in the North Sea in Collie *et al.* (2013) and Rochet *et al.* (2013). Metrics such as biomass and evenness showed that the release of fishing pressure did not rebuild the North Sea community to its structure and function when it was unfished. The reduction of fishing pressure has instead shifted the community structure towards a higher abundance of smaller planktivores and benthivores, including flatfish.

Species rebuild at different rates, and may therefore alter the trophic interactions in communities, which may cause hysteresis in community rebuilding, i.e. reaching an alternate stable state. Small changes may not lead to changes in the ecosystem structure and function, but large effects may jeopardise the achievement of management objectives. Sustainable exploitation implies that 'small' is precisely defined, and that data requirements to this end are provisioned. The bottom line of the examples above is that rebuilding fish stocks to unfished levels by conventional stock assessments is increasingly being challenged, and that there is a pressing need for ecosystem models that can respond dynamically to ecological and environmental conditions (Collie *et al.*, 2014, Mackinson *et al.*, 2009).

1.5 Discards as an integral part of ecosystem effects of fishing

1.5.1 Fisheries' discards and single species populations

Fishing mortality results from different sources of mortality: mortality from landing (part of) the catch, mortality from discards, mortality after escaping the fishing gear during the catching process, mortality from lost gears (ghost fishing), mortality from actively avoiding a fishing gear, and so on. Quantitative data are available in decreasing order from the mortality by landing the catch over discard and escape mortality to 'avoidance mortality' (Broadhurst *et al.*, 2006; He, 2010). Fisheries management has for long only considered the landings as a capital source of fishing mortality, hence the definition of Total Allowable Catches (TACs). TACs were defined as quotas for the landings, but recent initiatives attempt to move towards catch quota (Kindt-Larsen *et al.*, 2011; Kraak *et al.*, 2013). Catch quota account for the mortality from landings and discards, while other sources of mortalities remain unaccounted as quantitative estimates are extremely sparse.

As both landings and discards may contribute significantly to fishing mortality, their combined effects should be investigated (Bellido *et al.*, 2011; Guillen *et al.*, 2014; Mesnil *et al.*, 1996). Discard mortality, however, is not just an extension of the mortality from landings. Fishing is selective, and the mortality rates imposed to different sizes and species are different for the landings than for the discards, as both sources of fishing mortality have different causes. The distinction between discarded and landed species originates from the human perspective on fishing, which relates to either the economic interest of fishermen in making money or the societal interest in food provision and food security (Christensen & Raakjær, 2006; McClanahan *et al.*, 2015). Large species with high nutritional value are typically landed, while smaller species which are not of interest to human consumption are typically discarded. The process is further complicated by fisheries management, which attempts to safeguard fish for future generations imposing additional causes for discarding.

Differences in size and species composition of the discards and the landings have different effects on the stock and reproductive capacity (Fernández *et al.*, 2010). The assessment of the impact of fishing mortality on fish stocks requires, by consequence, data on landings and discards. Analytical stock assessments in 2011 were not available for 62% of the fish stocks in European waters due to a lack of biological information of the individual stocks coupled with inaccurate or missing catch data (Macdonald *et al.*, 2014a). The lack of discard data led to a growing tendency to quantify discards at sea over the last decades (Borges *et al.*, 2004; Rochet *et al.*, 2002). The focus was directed towards commercial (mostly TAC) species to improve the fish stock assessment models (Benoît, 1996).

Discards do not only differ significantly from landings by size and species composition, but also by their potential to survive the fishing process. Landings are more easily quantified and monitored as they are dead and extracted from the marine ecosystem. Discard data are more difficult to sample as they are not brought to shore, and they may, in contrast to landings, also survive the catching process and contribute to the biomass of the fish population in the sea, not only by surviving but also by increasing their own biomass and potentially producing offspring (Raby *et al.*, 2014a; Wilson *et al.*, 2014). Whether the contribution of surviving individuals can be high for single species populations remains unclear. This question was recently posed to an STECF-expert group of fisheries' scientists³, but their advice was limited to qualitative descriptions (Scientific Technical and Economic Committee for Fisheries (STECF), 2014a):

'The proportion of discarded fish that survive can be substantial. This depends on the species, the fishery and its operational characteristics e.g. gear type, tow duration as well as other technical, biological and environmental factors. Obliging fishermen to land catches of fish that would otherwise have survived the discarding process could, in some specific cases, result in adverse consequences for the stock. However, the choice to exempt a particular species based on "high-survival" is a "trade-off" between the stock benefits of the continued discarding of "high" survivors and the removal of potentially strong incentives to reduce unwanted catches by allowing discarding to continue. This should also be seen in the context of future stock benefits of improvements in selectivity on all species caught in the fishery as well as broader ecosystem benefits.'

Despite a limited number of studies on the survival of discards (STECF, 2014a) and limitations in the availability of discard data (Macdonald *et al.*, 2014a), discards are put forward as a substantial source of fishing mortality and should be accounted for in fisheries management (Bellido *et al.*, 2011; Kraak *et al.*, 2013). Discard data are therefore of paramount importance as the characteristics of discard mortality are significantly different from mortality induced by landings (Fernández *et al.*, 2010).

³ Survival of discards is highly topical in Europe, because Article 15 of the basic regulation of Common Fisheries Policy (EU, 2013a) provides an exemption from the landing obligation for 'species for which scientific evidence demonstrates high survival rates, taking into account the characteristics of the gear, of the fishing practices and of the ecosystem'.

1.5.2 Fisheries' discards and disruption of species interactions

The potential of population recovery after harvesting does not depend exclusively on fishing mortality (landings and discards) of a single species (chapter 1.4.1; chapter 1.5.1), but may also be influenced by changes in species interactions (chapter 1.4.2; this chapter). Landings affect species interaction by size- and species-specific mortality rates, and discards follow a similar mechanism.

Discards, however, have an additional and distinctly different impact in changing species interactions, as they are not extracted from the sea but instead, directly serve as food for scavengers across several trophic levels in the food web: birds, marine mammals, demersal fish, benthic invertebrates on and in the seabed (Figure 1.7).

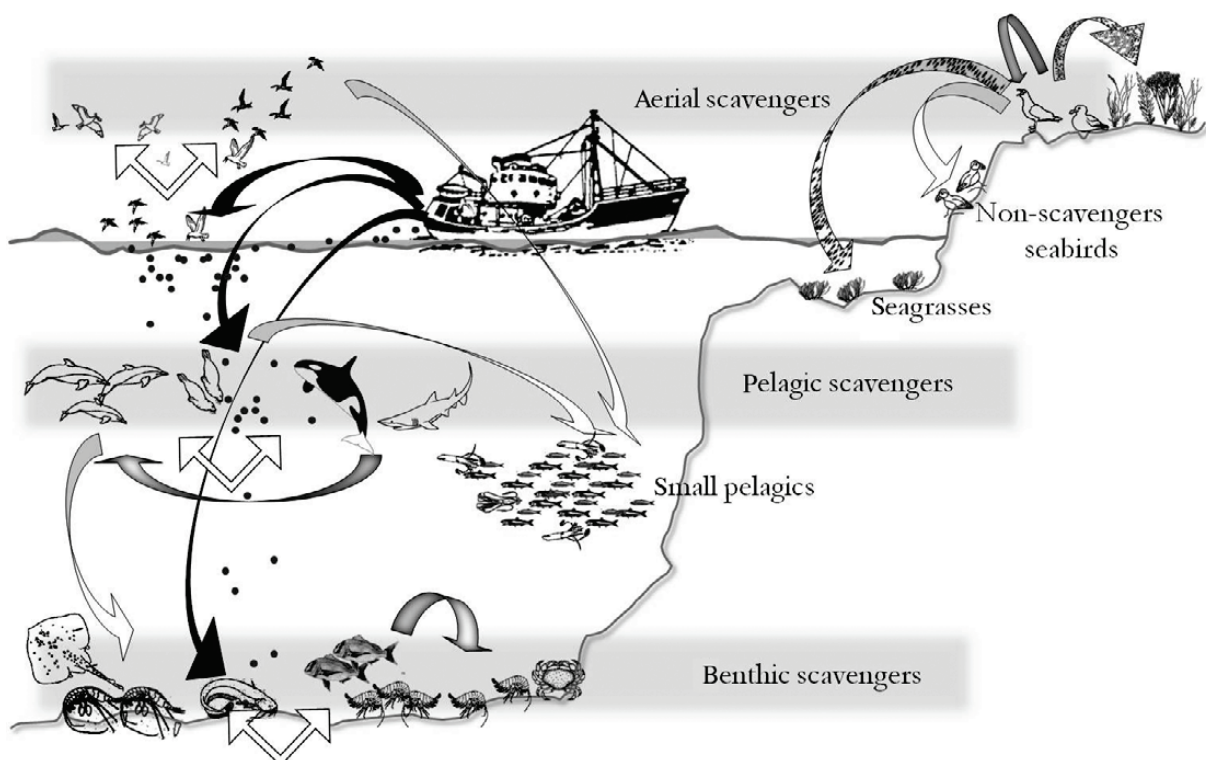


Figure 1.7 The effect of fisheries discards as an alternative food source includes five types of ecosystem interactions: (1) scavenging on discards by seabirds, meso-pelagic and benthic scavengers (black arrows), (2) predation within communities (shadow grey arrows), (3) resource and interference competition within communities (double white arrows), (4) hyperpredation or the increase in predation pressure from a generalist predator following the introduction of an alternative prey (white arrows) and (5) transfer of nutrients (stained arrows) (after Oro *et al.*, 2013).

Figure 1.7 conceptually illustrates how the additional food from discards may be exploited by a large number of scavenging organisms, from top predators to invertebrates, and covering several habitats from the sea surface to the seabed. Resource as well as interference competition occurs both within and across trophic levels, often exacerbating changes in nutrient flows (Oro *et al.*, 2013).

Fisheries' discards play a different role in changing the ecosystem structure than mortality-induced changes. A simplistic and unidirectional representation of the ecosystem structure may be given by a food chain from carnivores over herbivores to plants and nutrients (Figure 1.8). Fishing mortality typically reduces the abundances of certain components and cascades through the food chain in an attenuating way. The effect decreases with each successive trophic level (top-down cascade). The role that fisheries' discards play as a food source is more likely to lead to a bottom-up cascade. Discards change the nutrient supply across all or most trophic levels in the food web, leading to similar changes in abundances at all trophic levels (Heath *et al.*, 2014a; Heath *et al.*, 2014b; Viana *et al.*, 2013) (Figure 1.8).

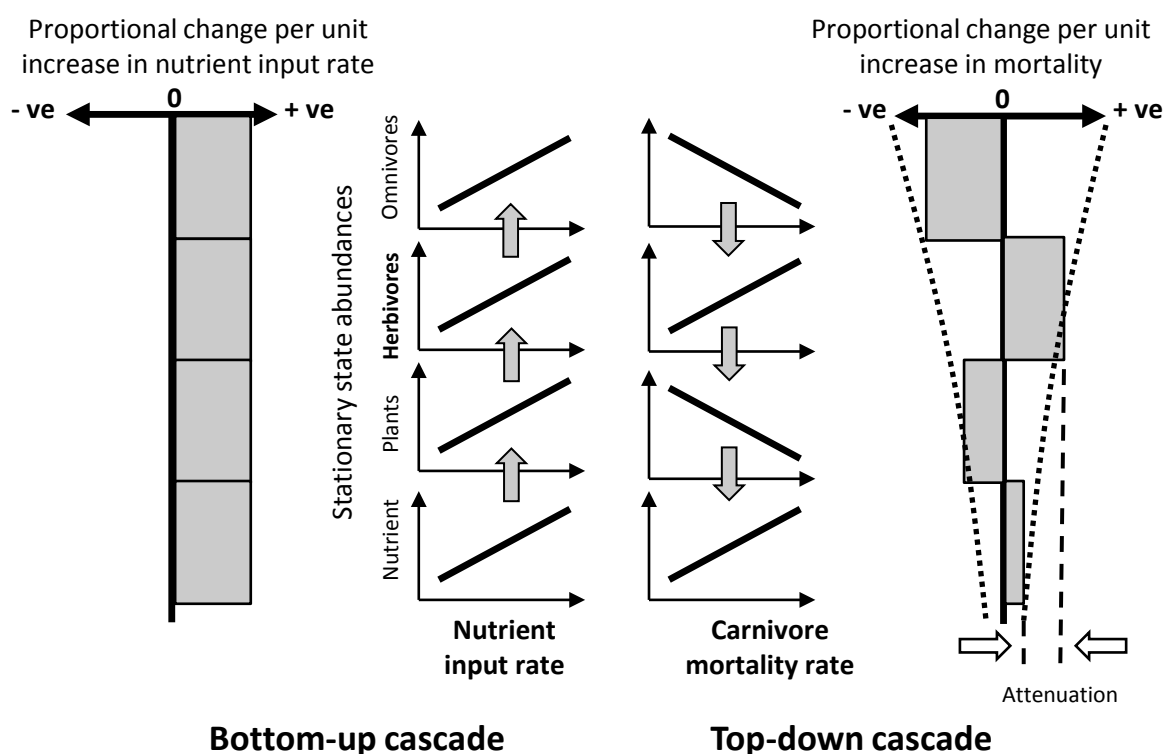


Figure 1.8 Theoretic and conceptual representations of trophic cascades. Bottom-up forcing leads to similar changes across several trophic levels, whereas top-down cascades typically result in opposite trends in abundances and attenuate further away from the driving force, e.g. top predators. 've' indicates deviations from the baseline situation, proportional changes to the variable prior to perturbation. A perturbation may for instance increase the natural mortality of herbivores (right graph), which implies an increase of mortality to the baseline level, as indicated by the proportional change '+ve' (after Heath *et al.*, 2014b).

The role of discards as a food subsidy has also been modelled for the North Sea and was shown to substantially affect different organisational levels of the ecosystem, although the approach was limited in the sense that it could not resolve effects at species level or foraging adaptations to exploiting alternative food sources. The implications of this modelling exercise are clearly that discards may subsidise not only seabird communities (Bicknell *et al.*, 2013), but also other

components of the ecosystem (Catchpole *et al.*, 2006; Furness *et al.*, 2007; Groenewold & Fonds, 2000; Kaiser & Hiddink, 2007). The specific role of discards as a food subsidy is not fully unravelled for different scavenging species populations, but has shown to affect their abundances and hence the ecosystem structures to significant levels. In general, predictable food sources from human activities, such as fisheries' discards, decrease the temporal variability of scavenging populations (McCann *et al.*, 2005) and increase their resilience and carrying capacity. Subsidised scavengers' populations may in turn reduce species diversity within communities and lead to a different structure and functioning of the marine ecosystem (Oro *et al.*, 2013).

1.6 Aim and objectives

Looking is not seeing.

What is self-evident from one perspective may not be seen so from another. The value of what is thrown away in the marine ecosystem should not be underestimated. Traditional fisheries management, notably the CFP, focuses on the sustainable extraction of commercial fish and shellfish species, and sees the discards of commercial fish and shellfish species as an unwanted source of mortality for these resources as well as a non-utilitarian waste. The objectives stipulated in fisheries management are inspired from this human and stock perspective.

The implementation of the ecosystem approach to fisheries management (EAFM) is increasingly prompting views from perspectives of other constituents of the marine ecosystem, such as non-commercial fish, seabirds, benthic communities, etc. Implementation in Europe is practically envisaged through the MSFD, although the policy agenda jumps ahead of science know-how, as a thorough understanding of the biology and ecology of the marine environment has a complex and highly dynamic nature (Dickey-Collas, 2014; Gascuel *et al.*, 2012; Jennings & Rice, 2011; Walther and Möllmann, 2014). The objectives of the MSFD are inspired from an ecosystem perspective and relate to the impacts of various human activities at sea, including fisheries.

Fishing effects such as discarding unwanted catches do not only affect commercial stock, but have significant impacts on non-commercial species as well. The consequences of discarding exhibit direct or indirect effects, depending on the species' role in the ecosystem structure and functioning. Effects range from (i) mortality after being discarded, over (ii) access to additional food resources from the discarding process to (iii) indirect effects from altered species interactions (predator-prey interactions and/or competition) (Fung *et al.*, 2015; Heath *et al.*, 2014a; Thrush *et al.*, 2015).

Assessing the effects of discarding on commercial stocks is at the core of European fisheries management (CFP) and must also be addressed under the provisions of the MSFD (Descriptor 3). This perspective is called the 'human and stock perspective' on discards in this thesis (Figure 1.9). Assessing the effects of discards on biodiversity, food webs and possibly seafloor integrity is also to be addressed within the MSFD. Assessing the effects of discarding on species with less or no value besides the effect on commercial stocks encourages the understanding of the role of discards in the ecosystem as a whole. The holistic approach leads to an ecosystem perspective, which serves aspirations for biodiversity conservation as well as sustainable use.

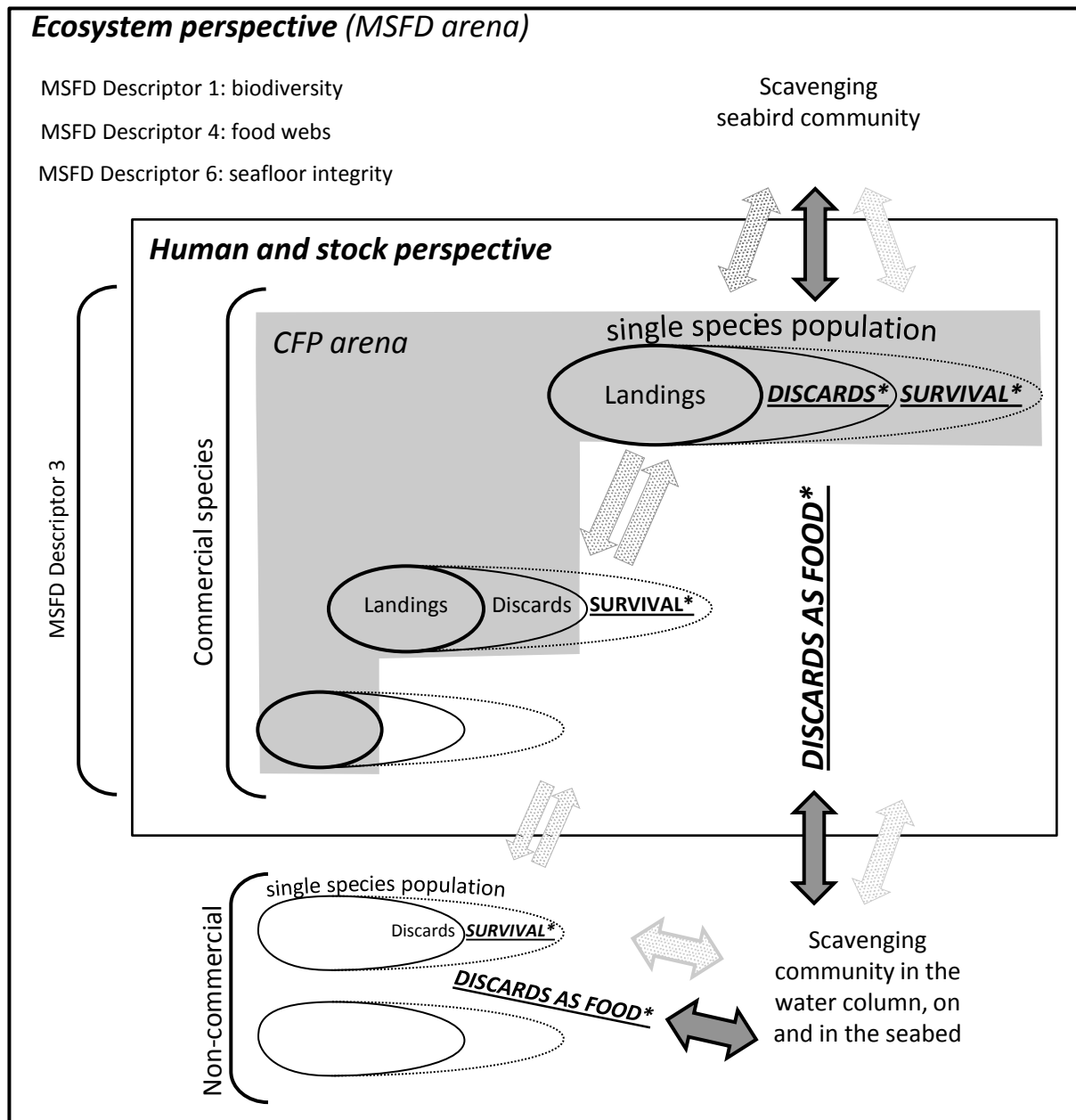


Figure 1.9 Fisheries' catches from the 'human and stock' and the 'ecosystem' perspective. Ellipsoid shapes indicate the fisheries' catches of commercial and non-commercial species. Species from the scavenging benthic and bird communities may also be comprised in the non-commercial catches. The catches of non-commercial species are entirely discarded (Ulleweitt *et al*, 2010), but some individuals may survive, whereas the catches of commercial species are also partially landed. The Common Fisheries Policy (CFP arena) emerged from the human and stock perspective which focuses up to today on commercial species, whereas the MSFD arena (Marine Strategy Framework Directive) also includes non-commercial species and the effect of fishing and discarding on ecosystem indicators such as biodiversity, food webs and seafloor integrity. Single arrows indicate the interactions between species (competition, predator-prey, and so on), whereas double arrows illustrate food relationships. The topics of this thesis are cross-cutting the human and stock perspective and the ecosystem perspective (bold capital letters* and dark grey double arrows).

The thesis looks into two perspectives on discarding, starting from the human and stock perspective with a pertinent focus on single-species populations (commercial stocks) and gradually moving towards an ecosystem perspective (Figure 1.9). Science is at the brink of investigating the ecosystem perspective by looking into the possibilities on estimating discard mortality of non-commercial species (quantification and survival) and by investigating the relationship between discards and other ecosystem components such as partitioning discards as a potential food source for seabirds or 'other' marine scavengers.

The thesis attempts to contribute to the human and stock perspective, where most of the focus is directed to, but also makes first attempts to quantify the importance of discards within the ecosystem perspective. While a holistic understanding cannot be realised within the realm of this thesis, the advances in constructing the building blocks aim to provide small pieces of the puzzle which may further stimulate research and management plans that integrate the role of discards as a source of mortality as well as a food source which may or may not subsidize other ecosystem components and alter the size and species composition of communities of several trophic levels and as such affect the sustainable use of marine resources and biodiversity conservation.

Research priorities to support a sound discard policy

Discards play a dual role in the ecosystem. They constitute a substantial source of fishing mortality (-) but also subsidize scavenging populations (+) which increases their resilience and reduces the variability of their abundance over time.

A first research priority is the collation of a comprehensive set of accurate and precise information on the quantities and composition of the discards in fisheries catches. The impacts of policy regulations cannot be evaluated in the absence of a comprehensive set of information on the quantities involved.

A second research priority relates to the implication of fisheries' discards as a source of mortality to species populations. Fisheries' discards are to be included in fish stock assessments to enable sound decision making on the exploitation rates of (commercial) fish populations. Discards add two major sources of complication which are less apparent in landings: (1) the difficulty to quantify discards (first research priority) and (2) the potential survival of discards (second research priority).

A third research priority relates to the functionality of discards in the ecosystem. On the one hand, discards induce mortality to commercial fish populations as well as mortality to non-commercial

species. These mortalities may cause trophic cascades throughout the food web, and ultimately alter community structures and functionalities. On the other hand, discards may also play a significant role in bottom-up forcing of the marine food web. The significance of this contribution to particular species groups is a research priority in fisheries management, as it may directly and indirectly affect (commercial) fish populations and compromise their sustainable use. The third research priority implies that the sustainable use of marine resources cannot be addressed as a stand-alone topic, i.e. the evaluation of single species populations requires that species interactions (including scavengers and discards) are accounted for to ensure sustainable exploitation of marine resources.

The fourth research priority evaluates the possibility to eliminate discards. Discards are unwanted as a source of mortality, because they do not provide food for humans, but instead compromise future food provisioning. Avoiding discards is therefore high on the priority list of fisheries' managers, fishermen and fisheries' scientists. Several discard mitigation measures have been developed, tested and assessed in various fisheries. The first, second and third research priority may become less relevant when discards can be eliminated.

Research strategy: aim and objectives of the thesis

The thesis is at the intersection of scientific studies within the current fisheries management regime and scientific understanding underpinning policy-support within the *'future'* implementation of the ecosystem approach to fisheries management (EAFM).

Fisheries' discards are an ideal case study, as they are an integral part of the core business in the Reform of the Common Fisheries Policy (current fisheries management). Discards play a dual role in the interaction of fisheries with the ecosystem (future EAFM). They are an additional source of mortality which may inflict changes in species populations and species interactions, depending on the amount of discards and the probability that they survive the discarding process, but discards are also an additional food source when they do not survive the discarding process or when they become available as an easy target for marine predators and/or scavengers.

The scientific studies in this thesis address the research priorities to support a sound discard policy by working:

'towards the quantification of the fate of discards from marine fisheries'.

The aim of the thesis was subdivided along the four research priorities (partims I-IV; Figure 1.10). The first partim is closest to the current fisheries management by addressing the quantification of

discards of mainly commercial species. The second partim is also directly related to current fisheries management by investigating the potential survival of commercially discarded fish species, but also makes aspirations to the ecosystem perspective (non-commercial fraction of the discards). The third partim investigates the possibilities to overcome discarding which is a key element in current fisheries management, while the fourth partim is furthest away from the CFP, but highly relevant in addressing ecosystem interactions. The fourth partim works towards the quantification of discards as a food source for other ecosystem constituents than humans.

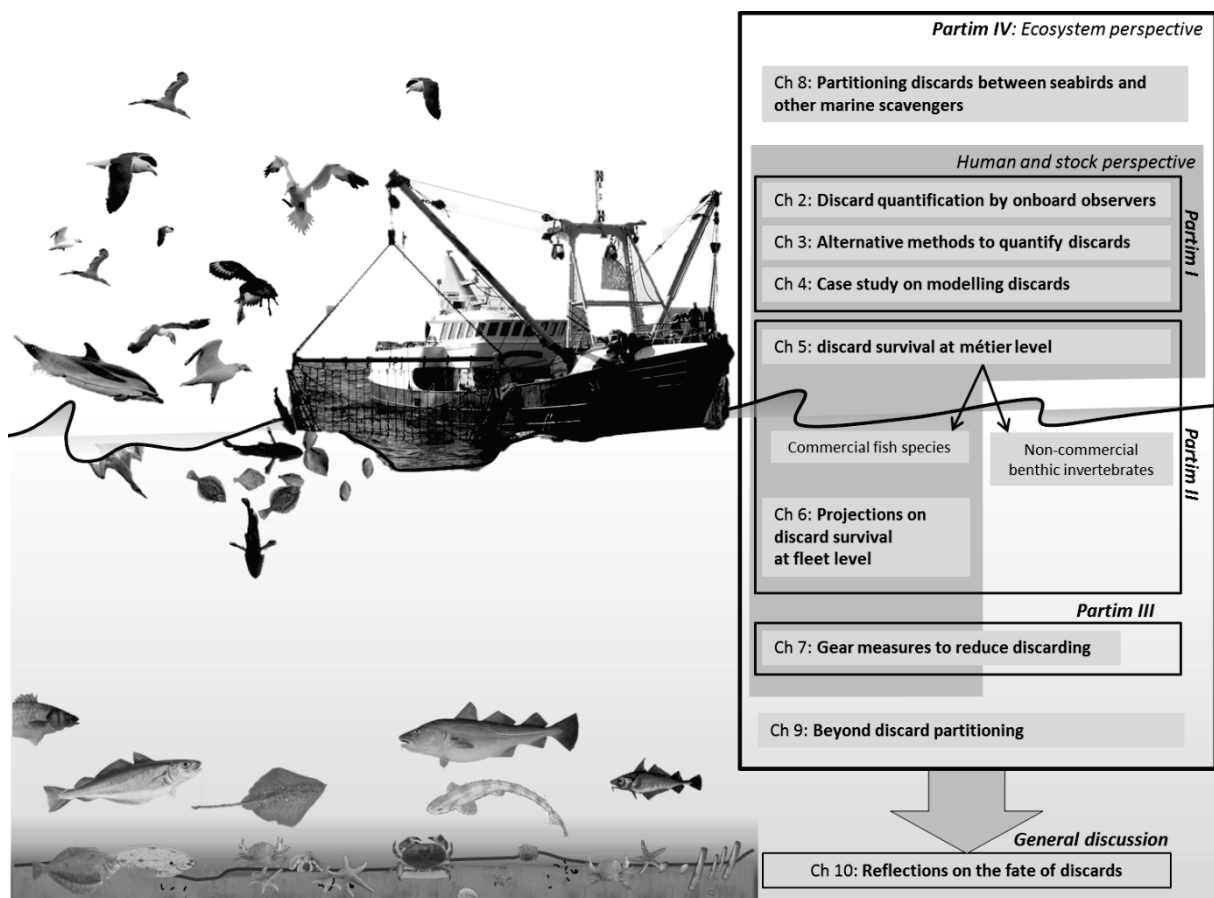


Figure 1.10 Outline of the thesis. Discards are generated by a fishing operation at the level of a fishing métier, i.e. a particular fishing gear targets commercial species in a specific location in time. The possibilities to quantify the discards for particular métiers are investigated in *Partim I*. *Partim I* includes a chapter on current estimation methods (Ch 2), possibilities to improve them (Ch 3) and a case study investigating the causes for discarding. The potential to survive the capture-and-discard process was addressed in *Partim II*, which includes a case study of in short-term survival in experimental conditions (Ch 5) and aspirations on how to evaluate survival at the full scale of the fishery (Ch 6). *Partim I* and *II* highlight the significant contribution of discards as a source of fishing mortality. *Partim III* evaluates the possibilities to avoid discard mortality (Ch 7). *Partim IV* integrates the ecosystem perspective by partitioning the amount of discards that are consumed by seabirds rather than marine scavengers under the sea surface. *Partim IV* also bridges the gap with the human and stock perspective (*Partim I, II, III*), as scavenging behaviour may compromise discard survival in the long term.

The aim of this thesis was addressed by the following specific objectives:

1. The human and stock perspective: towards the quantification of discards (Partim I)

Discard observer programmes and the use of the generated data were broadly evaluated, and solutions were listed to overcome the main issues. One of these solutions, modelling discards, was investigated in a case study on four commercial fish species in the Belgian beam trawl fishery in the southern North Sea. The case study was based on the identification of the main drivers for discarding, as a improved understanding of the variability in discard estimates may not only contribute to the direct quantification of discards, but also to the development of better sampling programmes and discard mitigation options.

2. The human and stock perspective: short-term survival of discards (Partim II)

The objective of the second study was to contribute to our understanding of the fate of fisheries' discards. The potential of discarded species to survive the capture and release process was investigated for commercial fish species and non-commercial benthic invertebrates in experimental conditions. Experimental conditions were fully representative of commercial practices, but could not cover the wide variety of environmental, technical and biological conditions of the entire fishing fleet. A secondary objective was the investigation of a practical way to evaluate short-term survival in a variety of fishing conditions.

3. The human and stock perspective: can we reduce discarding? (Partim III)

The Reform of the CFP has amplified its efforts to rebuild fish stocks to a healthy status by stimulating the development of measures that reduce unwanted mortality such as discards. Belgian research efforts to reduce discards have been intensified over the last decades. The objective of the third partim is the evaluation of the possibilities for Belgian beam trawl fisheries to eliminate discards using gear measures.

4. The ecosystem perspective: towards the quantification of the fate of discards (Partim IV)

The objective of this study was to partition the fate of discards into the share that is eaten by seabirds and the remainder, which is returned to the water. A specific objective of this study was to explicitly account for spatial and temporal variability in discard partitioning between aerial scavengers (birds) and scavengers in the water. The results of this study indicate the scavenging behaviour of seabirds on discards and thereby complement partim II on the survival potential of discarded organisms.

1.7 Outline of the thesis

This thesis is compiled from a wide range of information sources. The introduction sets the scene, based on recent advances in fisheries' science and the policy context of fisheries management. Chapter four, five and eight are stand-alone case studies and can be read in isolation. Chapter four and five are published as peer-reviewed publications, while the chapter eight was submitted for peer-reviewed publication. The other chapters are not published as such, but are based on a range of peer-reviewed publications in preparation, project and cruise reports, presentations, as well as desk studies conducted during ICES workshops and/or working group meetings. The linkage between the chapters is explained in the introduction and specifically demonstrated in Figure 1.10. The overall aim, moving towards the quantification of the fate of discards, was addressed by four lines of research. The research strategy is presented as four '*partims*' which tackle the priorities in understanding which role the return of discards may play in the marine ecosystem. The information sources of each partim are detailed below.

Partim I: Towards the quantification of discards.

This partim sets the scene. There is no estimation of the fate of discards if discards cannot be estimated. The estimation of discards, however, is not an easy task. Chapter two is a state-of-the-art on discard quantification and focuses in particular on the current estimates from discard observer programmes. Chapter two is a desk study using a wide variety of data sources which were internationally compiled in ICES or STECF working groups or in national data collection programmes. Information on discarded amounts and composition in Europe was partially based on a collaborative study within the ICES 'Working Group on Ecosystem effects of fishing' (WGECO) (ICES, 2014a). The WGECO-study was coordinated by the author and presented at the ICES Annual Science Conference in La Coruña (Spain) by María Fatima Borges (Borges *et al.*, 2014). Information was complemented with data from three national projects (IDEV, TOETS, WAKO-II), of which two projects were coordinated by the author: (1) IDEV: *Depestele, J., Polet, H., Van Craeynest, K., Vanderperren, E. 2008. Project 'Innovatiecentrum Duurzame en Ecologische Visserij'. Project VIS/02/B/05/DIV. ILVO-report 15p + Annexes* and (2) WAKO-II: *Depestele, J., Courtens, W., Degraer, S., Haelters, J., Hostens, K., Houziaux, J.S., Merckx, B., Polet, H., Rabaut, M., Stienen, E., Vandendriessche, S., Verfaillie, E., Vincx, M., 2012. An integrated impact assessment of trammel net and beam trawl fisheries (WAKO-II). Final Report, 233pp.* Conclusions were also partially based on a Master of Science thesis supervised by Prof. M. Vincx and the author: *Van de Walle, S., 2012. Subsampling accuracy in commercial beam trawl catches. Master of Science in Marine Biodiversity and Conservation, Ghent University.*

Chapter three focused on new development to estimate discards and was entirely based on literature review. It introduces chapter four, one possible new development to estimate discards.

Chapter four was based on a case study in the Belgian beam trawl fishery and focused on commercial fish species, as the Belgian observer program does not include all discarded species. The study was presented at the ICES Symposium on 'Fishery Dependent information' in Galway (Ireland) in 2010. The study was subsequently published in the Special issue of the ICES Journal of Marine Science, entitled 'Making the Most of Fisheries Information - Underpinning Policy, Management and Science'. This chapter is adapted from the published paper with reference: *Depestele, J., Vandemaele, S., Vanhee, W., Polet, H., Torrelee, E., Leirs, H., and Vincx, M. 2011. Quantifying causes of discard variability: an indispensable assistance to discard estimation and a paramount need for policy measures. – ICES Journal of Marine Science, 68: 1719–1725.*

Partim II: Short-term survival of discards.

Two chapters were included in this partim. Chapter five was based on a series of research campaigns on-board the Research Vessel 'Belgica'. Ship time was provided by the Belgian Science Policy Office (BELSPO). This chapter is adapted from the published paper with reference: *Depestele, J., Desender, M., Benoît, H.P., Polet, H., Vincx, M., 2014. Short-term survival of discarded target fish and non-target invertebrate species in the "eurocutter" beam trawl fishery of the southern North Sea. Fisheries Research. 154, 82-92.*

Chapter five specifically addresses the lessons drawn out of the case study in chapter four. The short-term survival of fisheries' discards is a research topic that yields highly uncertain and variable results despite being well-studied. A general review was not at stake, given the international research efforts within the ICES community (a series of consecutive workshops is addressing methodologies to estimate discard survival: WKMEDS). The author of this thesis contributed to two out three workshops (ICES, 2014c; ICES, 2015b). This chapter was based on initial discussions during the first ICES Workshop (ICES, 2014c) and the proposition in the discussion of chapter four on the development of a new survival proxy, the *Reflex Action Mortality Predictor* (RAMP). Preliminary investigations on the potential to improve the survival proxy from the case study in chapter four were addressed during a research campaign on-board the 'Belgica'.

The results were summarized in chapter five, which was largely based on the report: *Depestele, J., Buyvoets, E., Calebout, P., Desender, M., Goossens, J., Lagast, E., Vuylsteke, D., Vanden Berghe, C., 2014. Calibration tests for identifying reflex action mortality predictor reflexes for sole (Solea solea) and plaice (Pleuronectes platessa): preliminary results. ILVO mededeling nr. 158: 30p.*

Partim III: Can we reduce discarding?

Aspirations on the potential to manage discards are mainly focused on gear-related measures. The gear-related reflections are based on original work conducted by the author of this thesis in collaborations with fellow scientists at the Institute for Fisheries and Agricultural Research (ILVO).

The two main reports related to net modifications of beam trawls include original work by the author of this thesis, among others: *Depestele, J., Polet, H., Van Craeynest, K., Vandendriessche, S., 2008. A compilation of length and species selectivity improving alterations to beam trawls. ILVO-Report 56p.* and *Van Craeynest, K., Polet, H., Depestele, J., Stouten, H., Verschueren, B., 2013. ADVIS II: alternatieven voor de boomkorvisserij. ILVO-Mededeling 134. 172p.* Part of the work was conducted during the thesis work of two Master's students and one Bachelor student, supervised by Prof. M. Vincx and the author of this thesis: (1) *Goossens, J., 2014. De soortselectiviteit van boomkorren vergroten: hoe vangen we de doelsoort, tong (Solea solea) met zo weinig mogelijk teruggooi? Bachelor thesis, Gent.* (2) *Gorim, D.A., 2012. The potential of gear modifications to reduce the discards of beam trawls in the southern North Sea. Master of Science in Marine and Lacustrine Sciences and Management (Oceans and Lakes), Gent.* and (3) *Males, J., 2012. The potential of gear modifications to improve species selectivity of beam trawls. Master of Science in Marine Biodiversity and Conservation, Gent.*

Modifications to the catch stimulus of beam trawling to reduce fishing impacts were briefly described by literature search and one original study conducted by the author of this thesis and published in peer-reviewed literature: *Depestele, J., Ivanović, A., Degrendele, K., Esmaili, M., Polet, H., Roche, M., Summerbell, K., Teal, L. R., Vanellander, B., and O'Neill, F. G. Measuring and assessing the physical impact of beam trawling. – ICES Journal of Marine Science, doi: 10.1093/icesjms/fsv056.*

The potential to reduce the discards as well as other ecosystem effects by replacement of beam trawls with trammel nets was investigated in a national project coordinated by the author (WAKO-II) and published in peer-reviewed literature: *Depestele, J., Courtens, W., Degraer, S., Haelters, J., Hostens, K., Leopold, M., Pinn, E., Merckx, B., Polet, H., Rabaut, M., Reiss, H., Stienen, E., Vandendriessche, S., Volckaert, F.A.M., Vincx, M., 2014. Sensitivity assessment as a tool for spatial and temporal gear-based fisheries management. Ocean Coastal Manage. 102, Part A, 149-160.* The details of this study were not included in the thesis.

Partim IV: Towards the quantification of the fate of discards.

Chapter eight was based on a study conducted within the European FP7-project 'Benthis (grant 312088) and financial support of the Euromarine Mobility Fellowship. The study was reported in the Benthis project (Deliverable 4.5) and presented as a poster at a national (VLIZ Young Scientist's Day) and an international conference (ICES Symposium on the 'Effects of fishing on benthic fauna, habitat and ecosystem function' in Tromsø, Norway). Chapter eight was submitted to the Canadian Journal of Fisheries and Aquatic Sciences.

Chapter nine discusses the consequences of the partitioned discards. The first part elaborates on the consequences of discards as food for scavenging seabirds. The text is largely based on the results of WAKO-II project (Depestele et al., 2012), the collaborative study within the ICES Working Group 'WGECO' (ICES, 2014a) and the second ICES Workshop on estimating discard survival (ICES, 2015b), and a peer-reviewed publication co-authored by Jochen Depestele with reference: Sotillo, A., Depestele, J., Courtens, W., Vincx, M., Stienen, E.W.M., 2014. Discards consumption by Herring gulls *Larus argentatus* and Lesser Black-backed gulls *Larus fuscus* off the Belgian coast in the breeding season. *Ardea* 102, 195-205. The second part of chapter nine is largely based on advances during the ICES Working Group meeting of WGECO in April 2015 (ICES, 2015a) and tasks addressed in the Benthis-project. The results initiate the partitioning of discards to marine scavengers below the water surface. The epibenthic perspectives will be further addressed during the Benthis project and are coordinated by the author of this thesis.

Towards the quantification of discards

PARTIM I

Discards from human and stock perspective

2 Discard quantification by onboard observer programmes

There is no substitute for knowledge. – W. Deming

2.1 Introduction

European fisheries management is strengthening its discard management to improve stock status (Chapter 1). Management requires that the pressures from fisheries are well understood (Piet *et al.*, 2006), which implies that accurate information on the amounts and the composition of the fisheries catches, including landings as well as discards are needed. European Member States are legally bounded to collect data on fisheries catches as stipulated under the provisions of the Data Collection Regulation (DCR) (EC, 2000; 2001b) or its reformed formulation in 2008, the Data Collection Framework (DCF) (EC, 2008a; 2008b; 2009b), which will be replaced by the EU Data Collection Multi Annual Programme in new CFP (DC-MAP).

The DCF specifies the minimum requirements of national observer programmes to systematically collect fisheries data. National observer programmes were already established prior to the European legislation by several Member States, such as France, Ireland and Scotland (Borges *et al.*, 2005; ICES, 2000; Rochet *et al.*, 2002; Stratoudakis *et al.*, 1998). Regular observations of beam trawl fisheries' catches started from 2002 and 2004 onwards in the Netherlands and Belgium respectively (Van Densen & Van Overzee, 2008; Vandendriessche *et al.*, 2008). Discards in Belgian fisheries were thus systematically sampled from the starting date of the DCF. Data collection is likely to be recasted by the forthcoming changes of the 2013 Reform of the CFP, and not in the least by the landing obligation. This chapter, however, is based on the DCF up to 2014, because it is currently unknown what the practical implications of the Reform of the CFP will be for the at-sea data collection of fisheries' catches, how they will be implemented and what the output may be (EC, 2015).

The requirements of current discard observer programmes are described in detail in the DCF (EC, 2008a; 2008b; 2009b), which specifies how landings and discards are to be sampled. As observer programmes are financially demanding (Borges *et al.*, 2004), it may not be surprising that discard sampling is restricted. The first restriction applies to the metiers to be sampled. The definition of metiers is a prerequisite of the data collections. Metiers are defined as a group of fishing operations

targeting a similar (assemblage of) species, using similar gear, during the same period of the year and/or within the same area and which are characterised by a similar exploitation pattern (EC, 2009b). Only major metiers are included in the observer programmes, i.e. the most important metiers from an economic perspective. The 'most important' metiers are thus based on the landings and value of the commercial stocks and their fishing effort, but are complemented by metiers where discards exceed 10% of the total catch volume. The second, major restriction for DCF observer programmes relate to the species assemblage. Species that require sampling are defined in Appendix II of EC (2009), but may be 'summarized' as species commercially exploited in different regions and by different metiers.

The data collected under the DCF are supplied by each EU Member States as the basis for EU fisheries management. Fisheries' advice needs to be timely, accurate and reliable, and is largely composed by two advisory bodies, ICES and STECF. These advisory bodies evaluate whether discard data meet the scientific criteria to be used in stock assessment and directly serve fisheries' management advice on commercial stocks. The first objective of this chapter is to investigate the use of discard data in stock assessments and to highlight the role of Belgian fisheries in a selection of stocks that are commercially exploited by the Belgian fishing industry.

The DCF does not require systematic sampling of non-commercial species, and their quantities in fisheries' catches are thus largely unknown, except for those fisheries and national data collection programmes where all discarded species are registered. Discard data of non-commercial species is, by consequence, relatively sparse, and confined to isolated case studies (Lindeboom & de Groot, 1998; Shephard *et al.*, 2015). If the ecosystem consequences of discards are to be understood, detailed information on the composition and amounts of all discarded species is required.

Three selected case studies are presented in this chapter to evaluate whether the discards of non-commercial species are high in comparison to discards of commercial species. The first and second case study are based on discards from the French and Dutch fisheries, as sampling includes all discarded species (Fauconnet, 2014; Uhlmann *et al.*, 2013a). A similarly comprehensive data set is unavailable for Belgian fisheries and could not be established within the framework of this study, given its logistical and financial demands and scientific challenges. The total amount and composition of discards of commercial and non-commercial species have not been sampled in Belgian fisheries, but instead, a third case study will attempt to exemplify the importance of non-commercial species by presenting the numerical contribution of those species to the catches of several fishing trips by Belgian beam trawlers. This case study was based on data from various projects with another, though related aim, i.e. discard reduction (Depestele *et al.*, 2008b, 2012; Vandendriessche *et al.*, 2008).

The objectives of this chapter are summarized by the following questions:

1. Are discard data sufficiently accurate to be used in stock assessment and do they matter?
What is the fishing mortality (landings and discards) induced by Belgian fisheries?
2. What is the contribution of non-commercial species to fisheries catches and/or discards?

2.2 Materials and methods

2.2.1 Data sources

Investigations were mainly based on discard estimates from onboard observer programmes to fulfil data requirements of the European Commission Data Collection Framework (EC, 2008a; 2008b; 2009b) and retrieved from several international and national databases:

1. The STECF-database as produced by the STECF Expert Working Group in 2013 (STECF, 2013b).
2. Landings and discard data as reported in ICES stock assessment advice, hereafter called the ICES-database (ICES, 2015a; Available at <http://standardgraphs.ices.dk/stockList.aspx>).
3. The French onboard observer programme 'Obsmer' (Cornou *et al.*, 2013; Dubé *et al.*, 2012; Fauconnet *et al.*, 2011; Fauconnet, 2014).
4. The Dutch onboard observer programme (Uhlmann *et al.*, 2013a)
5. The Belgian onboard observer programme (Vandemaele *et al.*, in prep.; Vanelslander *et al.*, 2015).

The data were complemented by isolated catch data collected in the framework of several projects (IDEV, TOETS, WAKO-II) to investigate the numerical contribution of non-commercial species in Belgian fisheries' catches, as part of the second objective. A details on the data sources are described in Appendix 11.2.

2.2.2 Data analyses

2.2.2.1 Human and stock perspective: discards of commercial species

Discard observer programmes are restricted under the DCF. The index of Discard Coverage (DQI, STECF, 2013a: 105) denotes the landings for which discards are sampled (L_d , tonnes) and compares them to the overall landings (L , tonnes):

$$DQI = \sum L_d / \sum L$$

The DQI calculated from the STECF-database (DQI_{STECF}) is expressed by stock, fishery and Member State as the proportion of national landings covered by discard estimates in relation to the total national landings. DQI was also calculated for the ICES-database (DQI_{ICES}), albeit the calculation of DQI in this database is exclusively expressed by stock and not by fishery and member state as is the case for the DQI calculation using the STECF-database. The evaluation of the use of discard estimates in stock assessments was further based on the ICES-database using mean discard estimates for the years 2012-2014. These years were selected, because discard estimates were increasingly available in comparison to previous years (Appendix 11.1; Table 2.1). The history of using discards in stock assessments and management advice is briefly illustrated, and the relative contribution of discards to fisheries' catches (landings + discards) is illustrated for European fisheries as well as the Belgian part.

Discard rates were also summarized for the Belgian fisheries in the year 2012. Analyses were exclusively based on data from the Belgian discard observer programme as reported in detail in Vanelslander *et al.* (2015). The reported discard rate (D) was defined according to Rochet and Trenkel (2005):

$$D (\%) = (d / (d + k)) * 100$$

where d is the weight or numbers of the discarded fish (caught but not kept) and k is the weight or numbers of the harvested fish (caught and retained). These discard rates may differ from the discard rates that result from the ICES-database, as aggregations, raising procedures and/or inclusion criteria of observations may differ.

2.2.2.2 *Ecosystem perspective: discards of non-commercial species*

The discards of French fisheries are fully quantified by the Ifremer discard observer programme, i.e. the programme quantifies all discarded species (Cornou *et al.*, 2013). Species were split into species that did and that did not receive a TAC in 2012. Non-TAC species indicate the contribution of species with a limited commercial interest to the total discard composition in weight. The total discarded quantities of TAC and non-TAC species was compared.

The discards of beam trawl fisheries were fully quantified in the Dutch discard observer programme (Uhlmann *et al.*, 2013a). The species composition of the discards were presented for beam trawl fisheries in the North Sea in 2012, split up in small (≤ 221 kW) and large (>221 kW) beam trawlers.

The contribution of non-commercial species to the discards in Belgian fisheries was suggested from the catch composition of fishing trips with beam trawls in the southern North Sea and from the discard composition of two fishing trips in the English Channel (VIIdc) and the Bristol Channel (VIIf). Pie charts of the abundance of species or species groups in beam trawl catches or discards were compiled by region in order to illustrate the numerical diversity of fish taxa and taxa other than fish (benthic invertebrates as well as cephalopods). Taxa were grouped in the category 'other species' if they constituted $\leq 5\%$ of the total number of individuals. Taxa were identified to the species level or a higher taxonomic grouping (genus or family) if species identification was hampered by time or in case of doubt. The hauls were standardised by calculating the numerical abundance by hour (Catch/Discards Per Unit of Effort, CPUE/DPUE in number per hour). The diversity of fish and 'non-fish' species illustrated the number of species and their numerical contribution to the catch or discards. Qualitative inferences indicate the potential of non-commercial species to the discards of Belgian beam trawlers.

The catch of non-commercial species or species groups was subsampled, whereas the catch of commercial species was not. The accuracy of the subsampling procedure was evaluated from twelve hauls which were entirely analysed using 12 L buckets. The total catch was mixed and subdivided in 12 L buckets. Each of the buckets were analysed as if they were subsamples of the total catch.

The total abundance of each species in a haul was precisely and accurately determined by summing the numerical abundance of this species in all of the buckets. The abundance of the species was also calculated for different subsample sizes of the catches, based on one or a random combination of several buckets. The total number of individuals for a species is then based on the number raised by the weight of the subsample to the total catch weight of the haul. The abundance estimate using the total abundance of a combination of all buckets was compared to the abundance estimates using a

subsample of the catch. The comparison was based on the relative difference (sampling error E) in abundance from the total catch to the abundance of the raised subsample:

$$E = (A_s - A) / A \quad [1]$$

where A_s is the abundance as raised from the subsample and A is the total abundance as estimated from combining all buckets. This procedure was repeated with random combinations of buckets using >999 Monte-Carlo simulations in order to construct 95 % confidence intervals around the sampling error. Subsampling accuracy was then determined in function of different subsample sizes for four categories of taxa occurrences: rare, common, abundant or dominant. The categories of occurrence were based on the abundance index for each species in each of the hauls (Heales *et al.*, 2003):

$$N_i = 10 * T_{ij} / W_j \quad [2]$$

where $N_{i,j}$ equals the mean number of individuals of species i in a 12 L subsample, T_{ij} equals the total number of individual of species i in haul j and W_j the total weight of haul j (kg). Species were then grouped into four relative abundance categories. Rare occurrences were defined by an abundance index of ≤ 1 individual per 12 L sample, common from 1 to <5 individuals, abundant from 5 to <50 and dominant by an abundance index ≥ 50 . A similar procedure was applied to evaluate the number of species in function of the subsample sizes (see Van de Walle, 2012 for details).

2.3 Results

2.3.1 Human and stock perspective: discards of commercial species

2.3.1.1 Are the landings covered by discard estimates?

The STECF-database indicated that discard estimates were provided for <60 % of the total landed biomass from various fisheries in all regions except for fully documented fisheries (DQI_{STECF}) (Figure 2.1). This implies that discard estimates are only available for a limited number of fisheries. In contrast, discard estimates were available for a substantial proportion of the landings of a limited number of species for which stock assessment were available (DQI_{ICES} between 0.50 and 0.96; Table 2.1). The coverage was not reported for several other species.

The combination of both databases indicates that discard estimates are not available for a wide range of fisheries, but the coverage of landings is substantial for a limited number of species which are subjected to stock assessments.

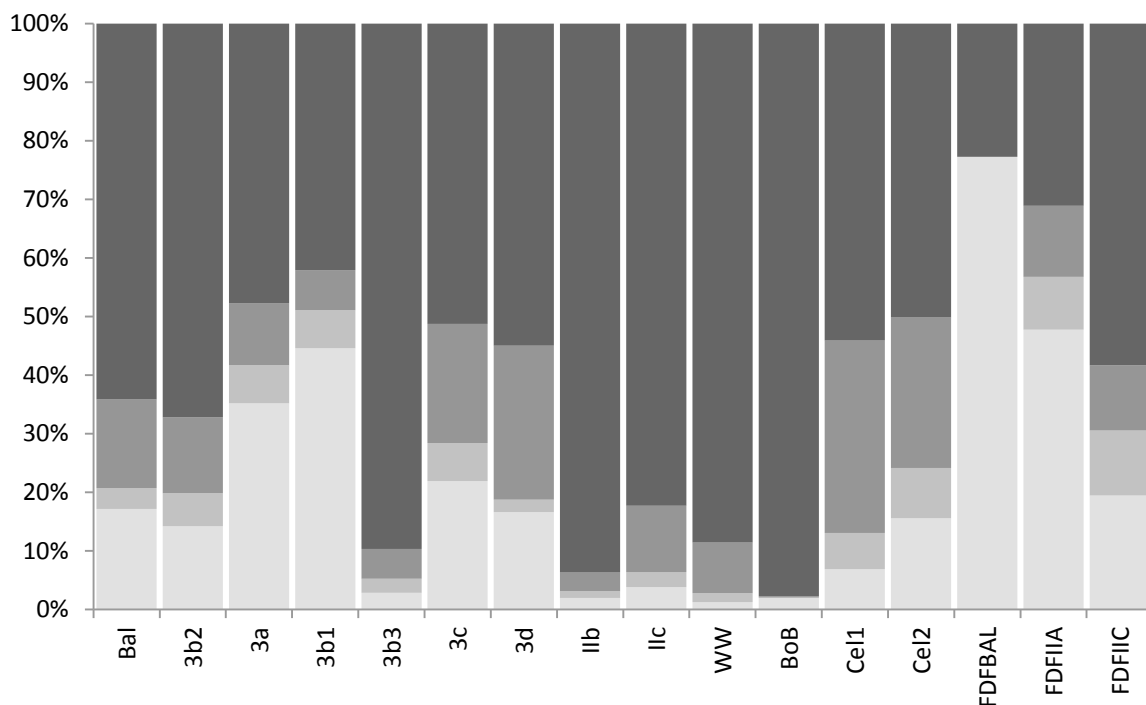


Figure 2.1. The percentage of each area (see Appendix 11.2 for abbreviations) for which the discards are sampled in relation to all landed biomass. Percentages are based on fisheries, defined by Member State, species, area and fishing gear. Some fishing gears are not sampled in certain areas, while others have been sampled only for a limited number of landed species. Greyscale relates to the DQI_{STECF} : from light to dark grey >67%, 34–66%, 1–33% and 0% (after Borges *et al.*, 2014; ICES, 2014a).

Table 2.1. Are discards used in stock assessments over time? Y: Yes, N: No. Stock codes are given in Table 11.2. Details on the discarded and landed biomass are provided in Appendix 11.1. Codes on how discards are included in stock assessments: 1: Full analytical assessment, discards included in the assessment (ICES stock data category 1); 2: Full analytical assessment, discards excluded from the assessment, but used for catch advice (ICES stock data category 1); 3: Age-based trends assessment, discards excluded from the assessment but used for catch advice (LPUE; ICES stock data category 3.2.0); 4: Survey based trends assessment, discards excluded from the assessment, but used to provide catch advice (CPUE; ICES stock data category 3.2.0); 5: LPUE trends based assessment, discards excluded from the assessment but used for catch advice (ICES stock data category 3.2.0); 6: Not included in the assessment and not included in the advice due to poor quality (ICES stock data category 3.2.0); 7: Not included in the assessment and not included in the advice (discards are considered negligible). Note that the proportion of the landings covered does not reflect the quality of the data.

	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	Proportion of the landings covered by discard estimates	How are discards included in stock assessment?
sol-nsea	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	0.96	1
ple-nsea	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	0.63-0.80	1
had-346a	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	0.9	1
cod-347d	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	0.69-0.83	1
whg-47d	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	0.73	1
lem-nsea	N	N	N	N	N	N	N	N	N	N	N	N	N	N	Y	Y	0.78	4
dab-nsea	N	N	N	N	N	N	N	N	N	N	N	N	N	Y	Y	Y	0.59	4
fle-nsea	N	N	N	N	N	N	N	N	N	N	N	N	N	Y	Y	Y	0.9	4
bll-nsea	-	-	-	-	-	-	-	N	N	N	N	N	N	Y	Y	Y	NN	5
tur-nsea	NN	NN	NN	NN	NN	NN	NN	NN	NN	NN	NN	NN	NN	NN	NN	NN	0.75	3
cod-iris	N	N	N	N	N	N	N	Y	Y	Y	Y	Y	Y	Y	Y	NA	NN	2
whg-iris	Y	Y	Y	NA	NA	NA	NA	NA	Y	Y	Y	Y	Y	Y	Y	NA	0.95	NA
sol-iris	N	N	N	N	N	N	N	N	N	N	N	Y	Y	Y	Y	Y	0.50-0.79	2
ple-iris	N	N	N	N	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	0.92	3
had-7b-k	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	NN	1
anb-p 78ab	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	NA	6
ple-7h-k	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	N	NA	6
sol-celt	N	N	N	N	N	N	N	N	N	N	N	N	N	Y	Y	Y	NN	2
ple-celt	N	N	N	N	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	NN	4
whg-7e-k	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	NN	1
cod-7e-k	N	N	N	N	N	N	N	N	N	N	N	Y	Y	Y	Y	Y	NN	6
ple-eche	N	N	N	N	N	N	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	0.92	1
sol-eche	N	N	N	N	N	N	N	N	N	N	N	N	N	Y	Y	Y	0.91	2
sol-bisc*	Y	Y	Y	Y	N	N	N	N	N	N	N	N	N	N	N	N	NA	7

- : No stock assessment; NA: Not Applicable (e.g. The ICES Advice for cod and whiting in the Irish Sea: no directed fisheries, lowest possible catch advised but no new assessments); *: Discards are estimated as partial estimates for the French offshore trawlers fleet; NN: Not reported in the ICES Advice

2.3.1.2 Are discard estimates used in stock assessments and do they matter?

Discard estimates are increasingly becoming available for species that are assessed in ICES stock assessments, although the discard estimates were highly variable for certain stocks such as anglerfish in the Celtic Sea and Bay of Biscay (ICES, 2015a). The use of the discard estimates in formal stock assessments was, however, limited to a low number of commercial stocks with an analytical assessment, such as the stocks of sole, plaice, cod, haddock (*Melanogrammus aeglefinus*) and whiting covering the North Sea. The stocks of species like lemon sole (*Microstomus kitt*), dab, European flounder (*Platichthys flesus*, hereafter called flounder) in the North Sea did not have analytical assessments, but were based on survey trends. Discard estimates of those species were reported since 2012 but exclusively used to provide catch advice, not in the stock analysis itself.

The relative contribution of discarded biomass to fishing mortality is nevertheless substantial for certain stocks (Figure 2.2). The discard rates of roundfish were highly different across areas for stocks of the same species. The discard rates of cod and whiting for instance were >60% in the Irish Sea, while they are <30% in the Southern Celtic seas and Eastern English Channel. In contrast, the discard rates of flatfish species were less variable across areas (e.g. <15% for sole, 30-70% for plaice).

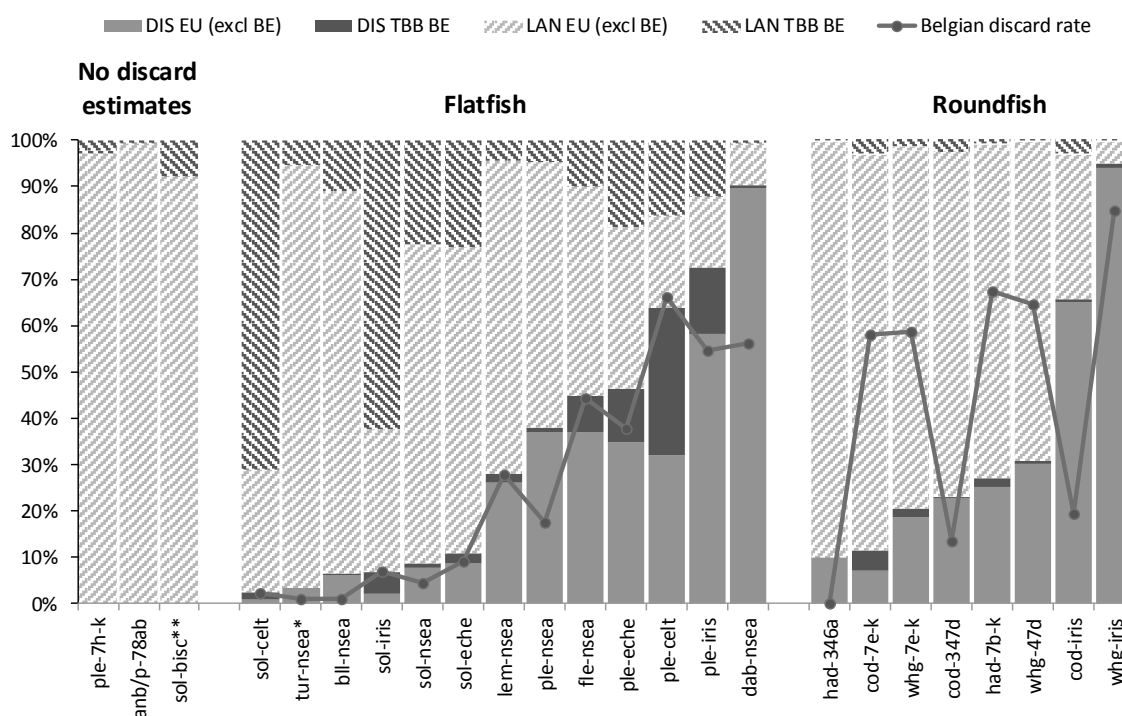


Figure 2.2 The relative contribution of discards to fishing mortality in stock assessment models for Belgian and European fisheries. Landings are indicated by the hatched bars while the plain bars represent the discards (dark: Belgium, light grey: EU member states other than Belgium). The discard rates (=discards/catch) for European fisheries are indicated in increasing order for flatfish and roundfish (gadoids). The Belgian discard rate is indicated by the grey line, which largely follows the European trend for flatfish but not for roundfish. LAN: landings, DIS: discards, TBB: Flatfish-directed beam trawl fisheries (mesh size: 80-99mm), BE: Belgium. The total biomass of the catches in each stock is given in Table 11.2.

2.3.1.3 *What is the contribution of Belgian discards to fishing mortality?*

The contribution of Belgian fisheries to the discards at European level is substantially different according to the fish morphology (Figure 2.2). The contribution of discarded biomasses from Belgian fisheries is marginal for roundfish (except for cod in the Southern Celtic seas and Eastern English Channel). The contribution of Belgian discards to the EU flatfish discards is, in contrast, substantial (e.g. plaice in the Celtic Sea and the Bristol Channel, ple-celt). The Belgian contribution to fishing mortality at European level is not surprising given its contribution to the EU landings of flatfish (e.g. sole in the Celtic Sea, Bristol Channel and Irish Sea).

Figure 2.2 also highlights that the discard rates of flatfish are largely similar in fisheries of Belgium and other Member States. In contrast, the discard rates of roundfish are lower for cod in the North Sea, Eastern English Channel and Irish Sea and higher for cod in the Celtic Sea and the Bristol Channel and whiting in the North Sea. Differences in discard rates of roundfish are caused by the dominance of other gears than beam trawls in the landings of these species. Beam trawling contributes for instance to 11% of the landings of cod in the Irish Sea, while otter trawls targeting *Nephrops* land 48 % of the whiting biomass. Beam trawlers (80-99 mm codend mesh size) in the North Sea account for 5 % of the cod landings, while the remainder is caught by gill nets, demersal (otter) trawls and seines (>100 mm) amongst others.

2.3.1.4 *How much do Belgian beam trawlers discard?*

Plaice and sole comprise the highest biomass of the landings in the Belgian beam trawler fleet (codend mesh size between 80 and 99 mm) (Appendix 11.2; Figure 11.2). Most of the landings originate from the North Sea (ICES Subarea IV) and the Celtic Seas (ICES Division VIIIfg). The overall discard rate does not vary much across the investigated fishing areas (Table 2.2), but the species contributing to the total discards vary considerably across areas (Figure 2.3). The overall discard rate was mainly determined by plaice in the North Sea (702 tonnes discards, 82 % of the total registered discards) and in the Irish Sea (106 tonnes, 87 %), pouting and dab in the Eastern English Channel (143 and 54 tonnes respectively, or 65 % and 25 %). The species that are contributing most to the discards in the Bristol Channel and Celtic Sea (ICES Divisions VIIIfg) are anglerfish (210 tonnes, 38 %), haddock (118 tonnes, 22 %), plaice (110 tonnes, 20 %) and lemon sole (85 tonnes, 15 %). Discards are, however, not registered for all landed species or there were too few observations to provide reliable discard estimates (Figure 2.3; Table 2.2). Note that the discards of plaice in the Eastern English Channel contribute up to 25 % of the European discards (Figure 2.2). The main discarded species by Belgian beam trawlers in the Eastern English Channel were, however, pouting and dab. The discard rates of Rajidae were between 24 and 42 %, which were not reflected from the ICES-database.

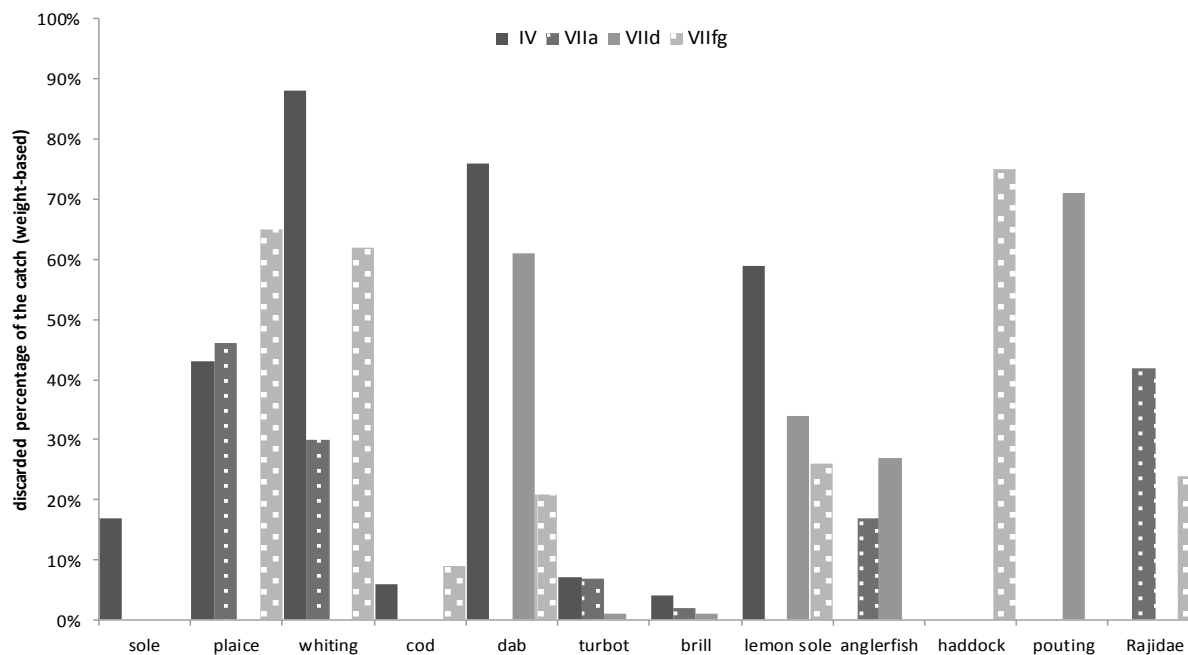


Figure 2.3. Weight-based discard rates in flatfish-directed beam trawl fisheries (80-90 mm codend) for four fishing areas. Discard estimates were not available for sole in VIIadfg; plaice and whiting in VIId; cod in VIId; dab and lemon sole in VIIa; anglerfish in IV and VIIfg; haddock in IV, VIIad; pouting in IV, VIIafg and Rajidae in IV and VIId (Based on Vanellander *et al.*, 2015).

Table 2.2. Landed and discarded biomass of Belgian beam trawlers in the North Sea (ICES area IV), Irish Sea (ICES Division VIIa), the Celtic Seas (VIIfg) and the eastern English Channel (VIId). The quantities of landings and discards only comprise the species which were registered during on-board observer trips (bold: registered; between brackets: landed but not registered). Species were ranked for each location according to decreasing weight of the landings. The top 10 or top 12 species in the landings were included, implying that species which are not landed may be under-represented (Based on Vanellander *et al.*, 2015).

Location	Species landed in highest quantities	Landings (t)	Discards (t)	discard rate
IV	plaice, sole , (flounder), cod, lemon sole , (dogfish), turbot, brill, whiting, dab	2465	860	26%
VIIa	plaice , (sole), (thornback ray), (blonde ray), monkfish , (dogfish), (cuckoo ray), brill, cod , (gurnards), turbot , (spotted ray), (Great scallop), undetermined rays	345	122	26%
VIIfg	monkfish , (sole), (megrim), lemon sole, cod , (blonde ray), plaice, haddock , (thornback ray), (Great scallop), dogfish, turbot, brill, whiting, undetermined rays	2039	549	21%
VIId	(plaice), sole , (cuttlefish), pouting , (Great scallop), lemon sole , (dogfish), turbot, brill , (monkfish), (gurnards), (dab)	630	219	26%
All locations		5479	1750	24%

2.3.2 Ecosystem perspective: discards of non-commercial species

2.3.2.1 Discards of TAC or non-TAC species in French fisheries

The discards of non-TAC species in French fisheries were at least of similar orders of magnitude in the Bay of Biscay, Celtic Sea and West of Scotland (Figure 2.4). The discards of non-TAC species were higher in the Western English Channel and in the Mediterranean Sea, but lower in the North Sea and the Eastern English Channel.

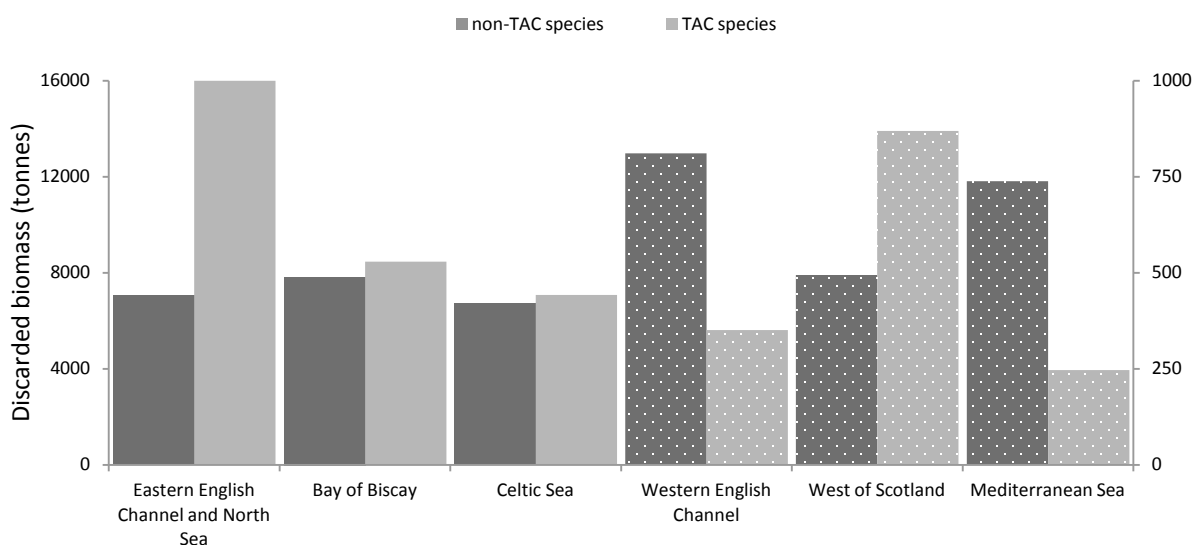


Figure 2.4. Discarded biomass (tonnes) of French fisheries in six fishing areas. The left (dark grey) bars indicate discards from species which could not be attributed to any TAC species. Light grey bars (right) include discards of TAC species (Modified from ICES, 2014a). Dotted bars refer to the secondary axis.

2.3.2.2 Species composition of Dutch beam trawl discards in the North Sea

The discards of Dutch beam trawlers in the North Sea comprise a wide variety of species (Figure 2.5). The number of fish species discarded by large beam trawlers (N=44) was higher than the number of species recorded in eurocutters (N=26). Scaldfish (*Arnoglossus laterna*) and solenette (*Buglossidium luteum*) were the most discarded species. The discards of benthic invertebrates were also highly diverse. A total of 75 species were recorded in large beam trawlers and 42 species in small beam trawlers. Echinoderms were discarded in the highest amounts, although the species differed. Large beam trawlers primarily discarded *Astropecten irregularis*, while *Asterias rubens* was the species with the highest abundance in discards of small eurocutters.

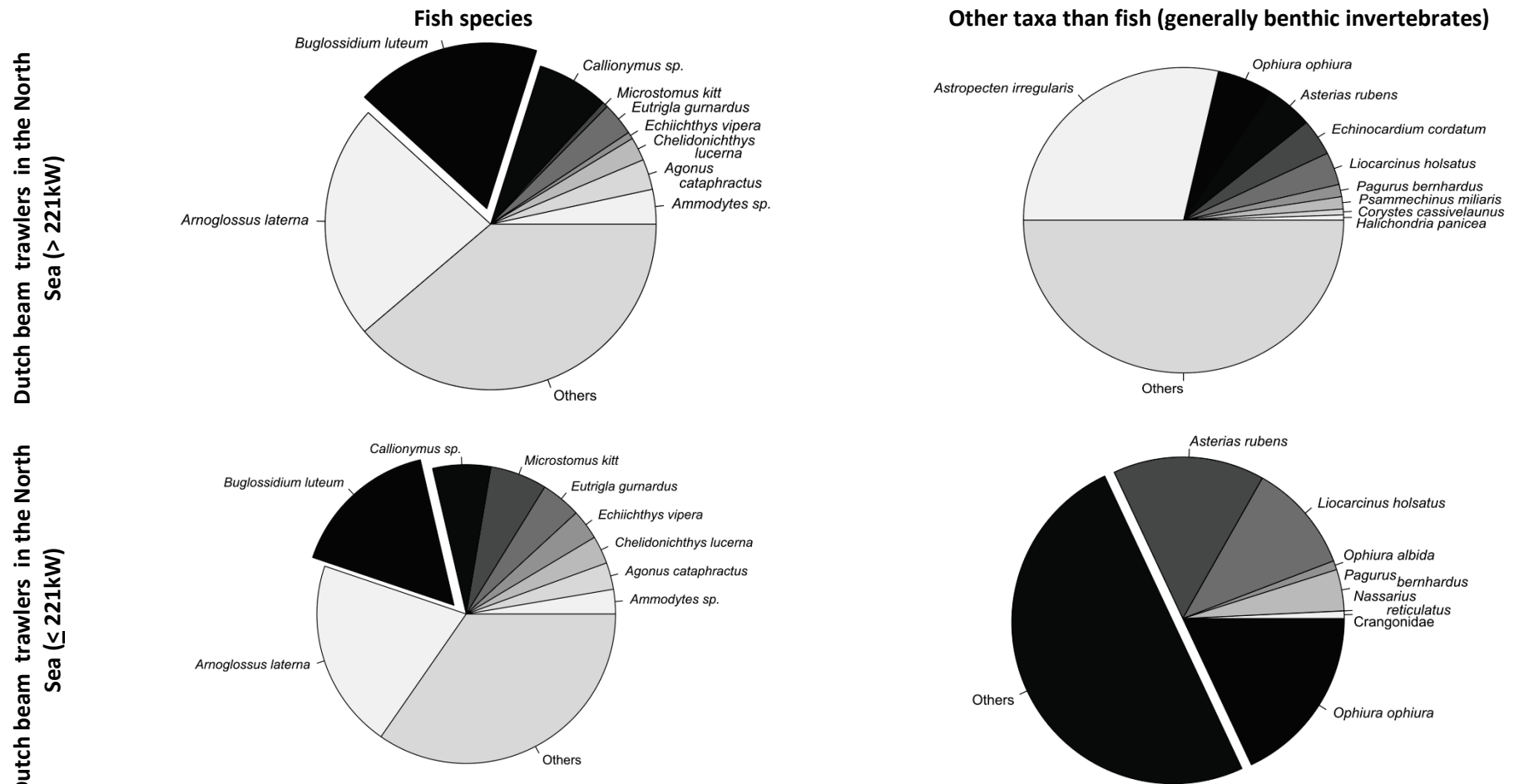


Figure 2.5. Mean number of individuals discarded by species in one hour beam trawling in the North Sea by Dutch beam trawlers ≤ 221 kW (lower rows) and large Dutch beam trawlers > 221 kW (upper rows) Total number of discarded fish is 558 and 3154 for other taxa than fish (lower rows); total number of discarded fish is 619 and 15448 for other taxa than fish, mainly benthic invertebrates (upper rows) (Based on data in Uhlmann *et al.* 2013a).

2.3.2.3 Species composition of Belgian beam trawl catches and discards

Catch and discard composition

The species composition of the beam trawl catches in the BPNS and the Western part of the North Sea was fairly different for fish, but was in both areas dominated by common starfish (*Asterias rubens*) for other taxa than fish (Figure 2.6). The discard composition of one hour beam trawling was considerably different in the Bristol Channel than the English Channel (Figure 2.7). Dab dominated the discards in VIIf, while discards in VIId were more diverse, comprising haddock, lemon sole, pouting amongst others. The total number of discarded individuals was also higher in the Bristol Channel than in the English Channel. Echinoderms dominated the discards in VIIf, while *Aequipecten* species were most discarded in the English Channel. About half of the catch in the English Channel consisted of fish, while <25% of the discards were fish in the Bristol Channel.

Sub-sampling accuracy

Two species dominated the catches of the twelve hauls in the Western part of the North Sea: *Asterias rubens* and *Psammechinus miliaris*. Abundant species included common whelk (*Buccinum undatum*), hermit crab (*Pagurus bernhardus*), flying crab (*Liocarcinus holsatus*), harbour crab (*L. depurator*), and so on. Common occurrence were hooknose (*Agonus cataphractus*), *Inachus sp.*, while sea snails (*Liparis liparis*), sculpin (*Myoxocephalus scorpius*) and the toad crab (*Hyas coarctatus*) rarely occurred.

Sampling errors were < 50 % when using subsamples of 12 L for dominant and abundant species. The relative difference between the estimated abundance by raising 12 L subsamples and the total abundance may be as large as 100% for common and rare species. The error rapidly decreases for dominant, abundant and common species when the subsample sizes increases (Figure 2.8).

The number of species recorded in small subsample sizes (as a fraction of the total catch) is about half the number of species recorded when the total catch was analysed (Figure 2.9). The number of species recorded increases rapidly when increasing subsamples from 10% of the catch to 20-30 %, but the benefits of increasing subsample sizes flattens when over half of the catch was analysed.

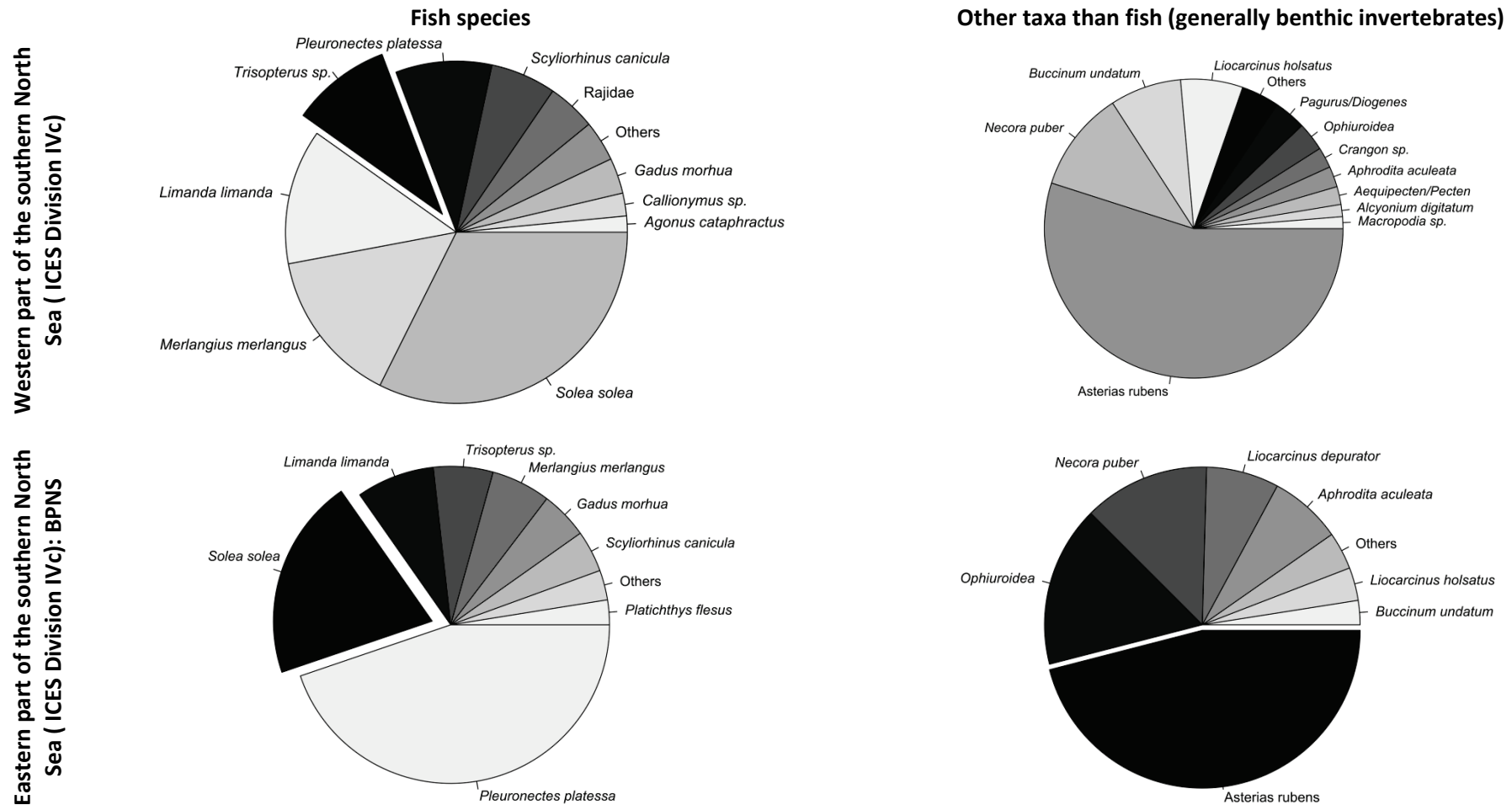


Figure 2.6. Mean number of individuals caught by species in one hour beam trawling in the Western part of the southern North Sea (total number of fish: 74, other taxa than fish: 891) and in the Belgian Part of the North Sea (total number of fish: 195, other taxa than fish: 637)

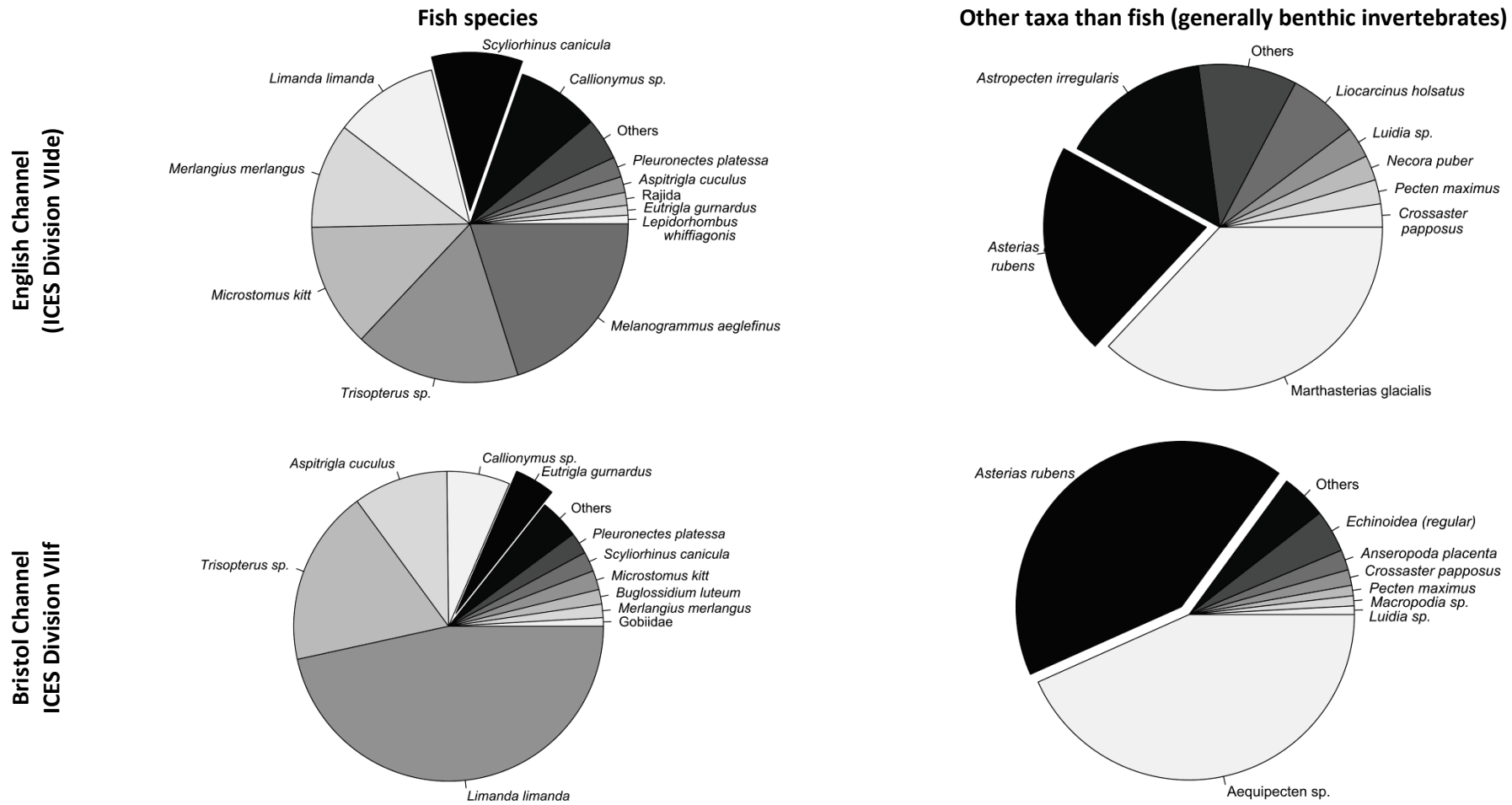


Figure 2.7. Mean number of individuals discarded by species in one hour beam trawling in the English Channel (total number of fish: 611, other taxa than fish: 584) and in the Bristol channel (total number of fish: 318, other taxa than fish: 2843)

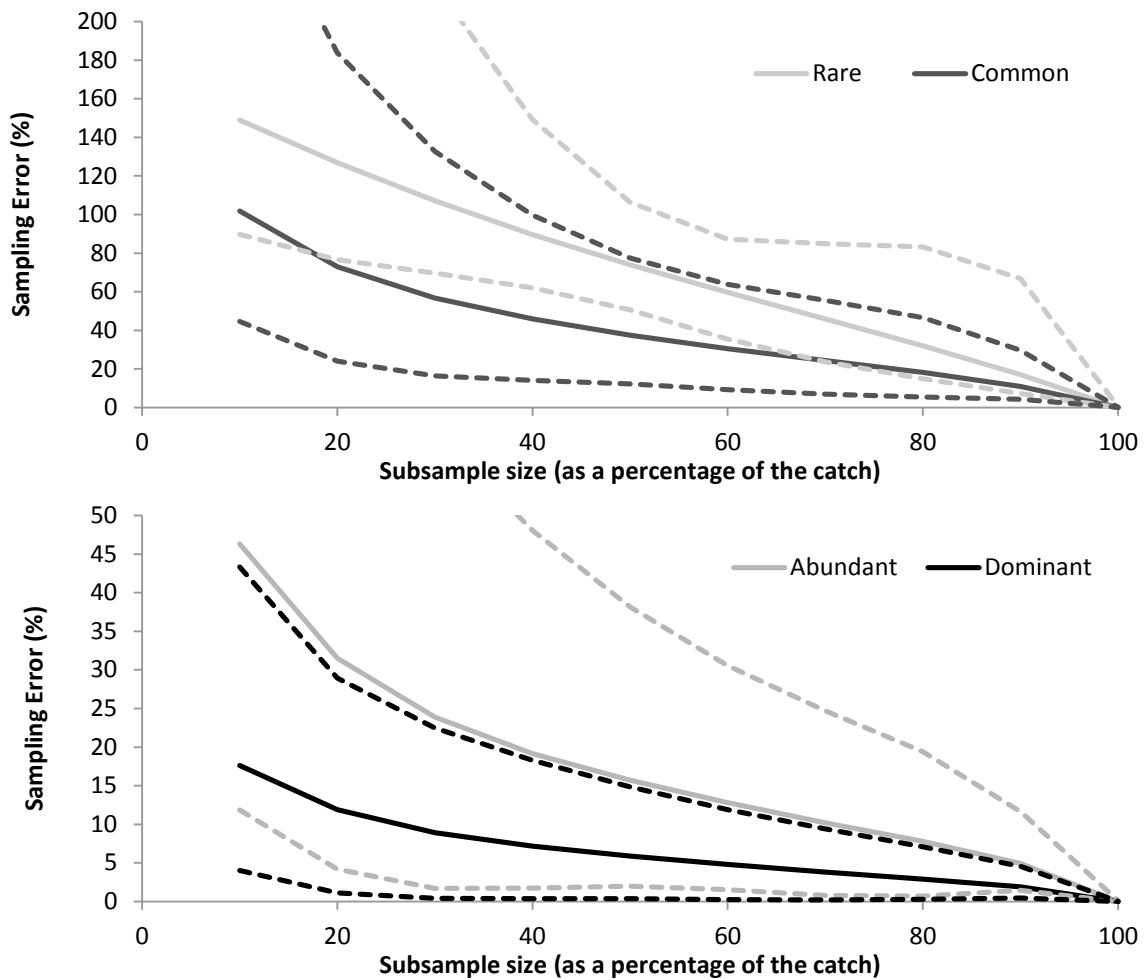


Figure 2.8. Mean sampling errors (full lines) and 95% confidence intervals (dashed lines) as a function of increasing sample sizes for abundant, dominant, common and rare species. Sampling errors were presented as percentage deviation of the observed numbers and are based on 12 hauls with a chain mat beam trawl in western part of the southern North Sea. (After Van de Walle, 2012).

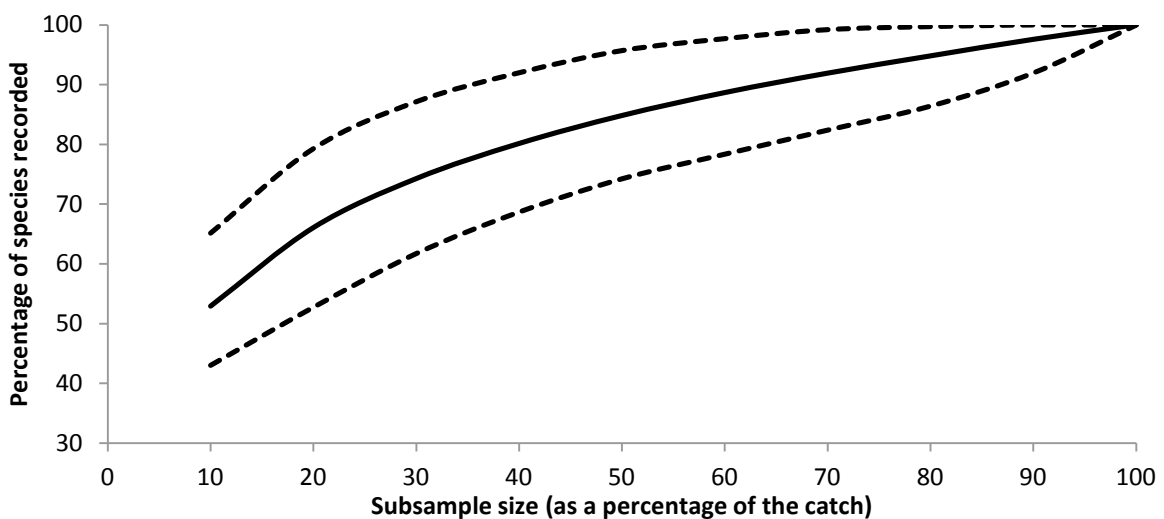


Figure 2.9. Mean number of species recorded and 95 % confidence intervals (dashed lines) as a function of increasing sample sizes, based on 12 hauls with a chain mat beam trawl in the western part of the southern North Sea (After Van de Walle, 2012).

2.4 Discussion

2.4.1 Discards from the human and stock perspective

2.4.1.1 Availability and use of discard data at European level

Discards have long not been included in stock assessments, but their contribution to fishing mortality was stressed in recent years (ICES, 2015b). The discards of sole contributed to <15 % of the fishing mortality, but discards of plaice in the Celtic Sea (VIIIfg) and in the Irish Sea (VIIa) contributed, in contrast to >60 % of the fishing mortality in the respective regions. Belgian discards were responsible for 50 % and 20 % of the European plaice discards in VIIIfg and VIIa. Fishing mortality of cod and whiting in the Irish Sea was nearly exclusively driven by discards. ICES advised to minimize the catches of both species in the Irish Sea and to stop any directed fisheries on either species (ICES, 2015a). Several North Sea fish stocks, including both the flatfish and roundfish stocks, are examples of high-value, well-studied stocks where the completeness of the data and assessments is high (including discards), as are the exploitation rates (Rice, 2011a; Trenkel *et al.*, 2015). The discards of whiting, for instance, in the stock covering the North Sea (whg-47d) was sampled in Belgian beam trawl fisheries (80-99mm codend) (Figure 2.3), but Belgian discard data were not included because of the discard data were not aged. Discard data were nevertheless available for 73 % of the landings of the whiting stock, based on data from France, Germany, England and Scotland. Whiting is of higher economic importance to fisheries of the latter member states such as the United Kingdom which have whitefish-targeting fisheries (Pope & Macer, 1996; Kerby *et al.*, 2012).

The reported discard data of those 'most important' commercial species generally covered a substantial part of the landings (>60 %), but the number of stocks for which discard data were reported was limited. Discard data were therefore only used in analytical stock assessments for a limited number of species.

The STECF database indicated that discard estimates covered <60 % of the EU landings for several fisheries. The availability of discard estimates in international databases (STECF, ICES) was limited or absent for many species and fisheries, generally the species with less economic value. The discards of those species, however, were generally higher than those of species with higher economic value and may constitute >90 % of the fishing mortality, such as for dab (Figure 2.3). Discard studies generally focus on the commercial most important species (Uhlmann *et al.*, 2014c; chapter 4), which is not surprising and may also be generalized to all fisheries data and analyses, including stock assessments (Baum, 2003; Dowling *et al.*, 2013; Salomon & Holm-Müller, 2013).

The investigations nevertheless highlighted that the contribution of discards to fishing mortality generally increased from high-value to low-value species in a specific fishery, but that, in contrast, the availability of discard data followed the opposite trend and decreased from high-value to low-value species. The usefulness of discard data may not be underestimated in stock assessments for species with limited economic value.

2.4.1.2 *Towards an international database of discards?*

The analysis of discard data across international and national databases was avoided in this study. Several examples illustrate why. A first example shows the discrepancy in discard estimates of the Belgian beam trawl fisheries using 80-99mm codends in the North Sea. Discard were estimated at 1372 tonnes in 2012 based on the ICES-database, while the estimates from the STECF-database indicated 1657 tonnes (Quirijns *et al.*, 2014). Another striking example shows that the STECF-database accounted for <600 tonnes discards in the Bay of Biscay, while the French national observer programme calculated a discarded biomass of about 16000 tonnes (Figure 2.4) (ICES, 2014a). A last example comprises the discard estimates of the 'unregulated beam trawls' in the discard atlas of the North Sea. The 'BEAM' metier link to the Belgian shrimp beam trawlers (Vanelslander, pers. comm.), explaining the limited amounts of plaice landed (19 tonnes). As discard rates for Belgian shrimp beam trawlers were not estimated in the Belgian discard observer programme, discard rates of neighbouring member states were used, which inflated the discards of plaice in the North Sea up to 8118 tonnes (Quirijns & Pastoors, 2014: 64).

The differences in discard estimates across databases may be a reporting issue as the example of the Bay of Biscay illustrated, but another, recurring cause is related to the sampling and raising procedures (Quirijns *et al.*, 2014). Discard estimates from different databases result from a complex process of aggregations and estimation methods for unsampled strata, which may be conducted following different methodologies. Discard data in the STECF-database, for instance, are based on aggregations of discard data by metier and fishing area from several Member States. Data gaps of one Member State are complemented by data from another Member State. Transferring discard rates across Member States may, however, introduce biases as the causes for discarding may differ between Member States (e.g. quota-regulations). Highgrading (or discarding of marketable) of plaice, for instance, is one of the examples where discard rates of plaice in the North Sea differ between Belgium and the Netherlands (Figure 2.2; chapter 4.5).

The Reform of the CFP has recently stimulated the joint presentation of national discard information. The implementation of the landing obligation requires discard management plans (EU, 2013a), and as stated in Article 14(2):

'Member States may produce a "discard atlas" showing the level of discards in each of the fisheries which are covered by Article 15(1)'.

A discard atlas is currently available for several regions and/or fisheries: the South Western Waters discard atlas (Rochet *et al.*, 2014a), the North Sea discard atlas (Quirijns & Pastoors, 2014), the North Western Waters discard atlas (Cefas, 2014), NWW pelagic and industrial discard atlas (Marine institute, 2014), and so on. These documents were based on a compilation of discard data from national discard observer programmes (Rochet *et al.*, 2014a) or on international databases such as the STECF-database (Quirijns *et al.*, 2014). However, this work has not led to international, regional discard databases. Discard data remained scattered in national databases or in the presented international databases of ICES and STECF, which serve specific purposes, such as stock assessments (ICES-database) or the evaluation of management measures (STECF-database). The high level of aggregation of the data hampers the use of discard data for other, detailed analyses, such as in chapter 8.

This study therefore recommends that a discard database is developed and made available at European level, which integrates discard data from national observer programmes and allows the use of discard data for purposes other than stock assessments or specific (fisheries) management measures. The current ICES and STECF databases are a step towards this type of integrated database, although they lack:

- Access for professional users other than those involved in the specific work, e.g. details on discard data in the ICES-database is restricted to stock assessment coordinators.
- Transparency in sampling procedures, such as the number of sampled trips (ICES, 2015b) and the number of individuals evaluated. A lack of information on sampling and aggregation procedures complicates the evaluation of the uncertainty (both variability and bias) in the reported estimates.
- Consistency in reporting, as raising procedures are for instance not always described. Sampling and raising procedures should either be clearly reported for each member state or observer programmes should be standardised (ICES, 2012b).
- Completeness, as discard information is restricted to the objectives of the database which restricts the scientific research opportunities from this source of information.

The provision of discard data to international databases should therefore specify discard procedures in a higher degree of detail, including specific discard sampling and raising procedures by member states. This information can to this date only be obtained by tracing national discard observation procedures, which is complex, time-consuming and sometimes even impossible. This lack of transparency in reported discard data hampers the interpretation of discarding effects at European level and requires considerable efforts if scientific advice on discards is to be given in a responsive way at regional and ecosystem level.

2.4.1.3 *Deployment and observation bias*

Several studies preceded the DCF in evaluating the possibilities to set up discard observer programmes that reduce the costs while obtaining accurate and precise estimates (Allen *et al.*, 2002; Borges *et al.*, 2004; Cotter *et al.*, 2002; Rochet *et al.*, 2002; Stratoudakis *et al.*, 1999; Tamsett *et al.*, 1999a, 1999b). The DCF has followed several recommendations and restricted sampling to the main commercial species and métiers (see Chapter 1 'Introduction'). Another key element of the discard observer programmes is the stratified sampling design whereby a fishing trip is the primary sampling unit, from which discards are raised to estimates at métier-level (Allen *et al.*, 2002; ICES, 2007c).

Several sources of uncertainty, however, remain in the methodology. These uncertainties are partially inherent to the proposed stratified sampling strategy, but also partially due to unrealistic objectives of the DCF. The DCF requires, for instance, that sampling intensity is proportionate to the relative effort for the species and métiers to investigate. A consequence of this requirement is, for instance, that most observer programmes only obtain between 1 % to 5 % coverage at population level (Rochet *et al.*, 2014a; Vanellander *et al.*, 2015). The low sampling coverage results in a lack of precision for discarded weights. Precision can be expressed by the Coefficients of Variation (CV) which is often above the minimum DCF requirement (20 %) (Jardim & Fernandes, 2013).

Obtaining precise and accurate discard estimates at fleet level is difficult because of (i) the inherently high variability of catches and (ii) limited sampling effort due to high costs (Needle *et al.*, 2015). All of the reported discard estimates in this chapter were potentially due to these sources of uncertainty, but none of the details of the programmes could be evaluated as data quality and validation are often underreported or inconsistently reported. The lack of reporting correlations between auxiliary variables and discards in using ratio estimators to raise discards to fleet level is an example of this underreported validation of raising procedures (ICES, 2013b). The main sources of bias occur because of the stratified sampling design and the sampling procedures within a stratum.

Deployment bias: is sampling stratification representative of the fishing activities?

European discard observer programmes do generally not reach full observer coverage, in contrast to some ITQ-managed fisheries in the US (Branch *et al.*, 2006). Discard data are collected for a representative part of the fleet and then raised to fleet level. Two types of complications arise.

The first complication relates to the representation of the fleet. The DCF (EC, 2009b) specifies that:

'the sampling intensity shall be proportionate to the relative effort and/or the variability in catches of the metier. The minimum number of fishing trips to be sampled shall not be less than 2 fishing trips per quarter.'

The fleet as a group of vessels may be relatively stable through time, whereas their activities, as reflected by metiers, may change over the years (Ulrich *et al.*, 2012, Vanellander *et al.*, 2015). Stratifying the fleet into vessels with similar characteristics is easily manageable by gear type and mesh size among other vessel characteristics. Fleet-based stratification allows observers to plan their sampling trips relatively well in advance, but this stratification has the disadvantage that the fleet activities and catch profiles may diverge by target assemblage, time and location. Catch profiles of fishing trips by gill netters, for instance, were correctly forecasted in >80 % of the trips, whereas the percentage of correctly forecasted catch profiles of demersal trawlers varied between 5 and 60 % in the French fishing fleet (Marchal, 2008). Stratification by fishing activity, which includes the target assemblages and potential variation in spatio-temporal patterns, is de facto only possible *a posteriori*. Metier-based stratification may slightly improve CVs and increase the sampling coverage (Cornou *et al.*, 2014).

The second complication of the stratification relates to the raising procedures. Estimating discards for a proportion of the population (fleet or metier) requires that discards estimates are raised to the population level. Common practice involves the use of an auxiliary variable as proxy for the fishing activity (ICES, 2007c; chapter 8). Ratio estimators may improve the precision of the raised discard estimate in comparison to simple random estimators⁴, but require a significant correlation between the discard estimates and the auxiliary variable (Allen *et al.*, 2001; Rochet and Trenkel, 2005). These correlations are generally not estimated and may introduce bias. Possible auxiliary variables include

- The landings of single species, target species assemblages or all species. Bias may be introduced by low landings or variations in target species. The high discard estimates of dab, pouting and flounder may be prone to this type of bias (Figure 2.3; Stratoudakis *et al.* 1999),

⁴ Measuring fishing effort in numbers of the sampling unit rather than any auxiliary variable using simple sampling theory, and raising the discard estimates by sampling units.

- Fishing trips. Bias may be introduced due to different trip durations),
- Fishing effort. Bias may be apparent for passive gears, as the gear may fishing while the vessels are in the harbour,
- Days at sea.

Observation bias: is sampling within a stratum biased by the observers?

Whereas deployment bias results from the non-random distribution of observers across sampling units (fishing trips), observer bias may result from

- Changes in fishing practice, catch utilization and location when observers are onboard of fishing trips (Benoît & Allard, 2009)
- Limitations in the selection and the number of sampling trips that can be conducted within a stratum. Selection of sampling vessels and trips depends on the willingness of the skippers to cooperate in scientific surveys as well as their capacity to put observers to sleep, and so on. The practicalities and logistics of reaching fishing vessels when they come to port in distant places may also further complicate the observer programme (Palmer *et al.*, 2014).
- The sampling procedures of the onboard observers.

The first two types of observer bias, non-representative observations and a limited number of observer fishing trips, may result in significantly different and highly variable discard estimates across fishing trips within a stratum. Highly variable discard estimates may be a plausible explanation for not using discard estimates in stock assessments (Figure 2.2), which cannot be resolved unless more trips are sampled. When the number of sampling trips is low, it is impossible to discriminate between natural variation in discarding and systematic errors. The high variability may be due to the natural variation of species' abundances at certain locations and times of the year, but may also be due to the stratification, e.g. fleet-based strata of demersal trawlers (see above: deployment bias).

The last type of observer bias, the sampling procedure, may also lead to systematic, but undetected errors. The relative size of a subsample in a haul, for instance, affects the sampling error of the true abundance of this species in that haul. From this point of view, species which occur in low numbers should preferably be sampled exhaustively in a few hauls rather than being monitored in small subsamples in many hauls (Figure 2.8; Figure 2.9; Vigneau *et al.*, 2007 in ICES, 2007c). Between-haul variability may on the other hand also affect discard estimates of low abundant species. The Belgian discard observer programme focuses on species which occur in high numbers. The discards of low abundant species may thus not be accurately sampled.

2.4.2 Discards from the ecosystem perspective

The case studies illustrated that the discards of species which contribute less or not to the economic importance of the fishery (non-TAC species, species with less or no commercial value) can be as high as or higher than the discards of commercially important species in both weight and numbers. The discarded number of species and individuals were highly variable across fishing trips, areas, beam trawls (Dutch tickler chain versus Belgian chain mat beam trawls), sampling methodologies, and so on. While this chapter does not discriminate between the plausible causes of this high variability, it illustrates that discards cannot be confined to the most important commercial species. The high variability in discarded species and amounts may be complemented with other case studies of Belgian, Dutch and English beam trawl fisheries (Fonteyne & Polet, 2002; Rasenberg *et al.*, 2013; Revill & Jennings, 2005; Van Marlen *et al.*, 2011a) and other demersal fisheries (Damalas & Vassilopoulou, 2011; Catchpole *et al.*, 2013).

The diversity and amounts of discards of both commercial and non-commercial interest affect the marine ecosystem in a variety of ways (see Chapter 1 'Introduction'). The MSFD is the practical implementation on the management of the marine ecosystem. Discards may affect at least three MSFD Descriptors besides the commercially exploited stocks (D3): Biodiversity (D1), food webs (D4) and seafloor integrity (D6). The Good Environmental Status (GES) of these three descriptors are jointly assessed in Belgian waters, and are specified by 17 environmental targets. Several targets are directly affected by discards, such as obtaining a positive trend in the number of thornback rays (*Raja clavata*) and in the median colony/body size of *Buccinum undatum*, *Alcyonium digitatum* and others (Dupont *et al.*, 2014).

As the Belgian discard observer programme is primarily focused on achieving the DCF-requirement, two major shortcomings may be highlighted to evaluate the effects on discards within the MSFD:

1. The objective of the Belgian onboard observer programme is focused onto the species that fall under the specifications of the DCF. The discard data therefore lacks discard data on species which are of importance in the MSFD.
2. The Belgian onboard observer programme is directed towards the status of the commercial stocks and operates at a lower spatial scale than is required for the MSFD. The number of sampled trips in the BPNS are limited (Figure 4.1) and the selection of métiers in the observer programme may not sufficiently cover the fishing activities in the BPNS. Chapter 4 investigated the possibility to predict discards in the BPNS by investigating the causes of discards of the commercial species in the Belgian beam trawl fishery.

2.5 Acknowledgments

The investigations were facilitated by Sofie Vandemaele, Kelle Moreau, Sofie Nimmegeers, Els Torreele, Sebastian Uhlmann and Bart Vanelslander and the ICES Working Group on Ecosystem effects of fishing (WGECO), as well as the assistance of ship owners, crew and onboard observers in the fisheries' observer programmes. Further support was provided by the Flemish Government and the European Commission (Data Collection Framework).

3 Alternative methods to quantify discards

It is impossible to estimate the quantity of small fish that is destroyed since it is impossible to estimate the amount that is shovelled overboard, dead or dying. – Holt (1895)

3.1 Introduction

Quantification of discard composition and discarded amounts is difficult, leading to imprecise estimates with a great range of plausible explanations for its variability (Chapter 4; Rochet & Trenkel, 2005). A good understanding of the sources of variability in fisheries discards increases our abilities to monitor, quantify and manage them.

The scientific community continues research efforts in this field, which may be categorized in two groups: (1) efforts to improve the methodologies to quantify the discards and (2) efforts to quantify the causes of variability. Investigations focusing on an improved understanding of the causes of discard variability are generally also proposing discard mitigation measures (Enever *et al.*, 2009; Morandeau *et al.*, 2014). The envelope of identified drivers of discard variability are primarily related to fishing métiers, the discarded taxa and a spatio-temporal component capturing a variety of drivers such as biological, environmental and socio-economic characteristics (Chapter 4, Catchpole *et al.*, 2013; Feekings, 2012; Feekings *et al.*, 2012; Feekings *et al.*, 2013; Macdonald *et al.*, 2014b; Madsen *et al.*, 2012; Morandeau *et al.*, 2014; Pennino *et al.*, 2014; Sigurardóttir *et al.*, 2015; Tsagarakis *et al.*, 2013; Tsagarakis *et al.*, 2012; Uhlmann *et al.*, 2014c).

A detailed overview of the causes of discard variability can be found in Feekings (2012: 25-35), and were not repeated here. The focus of this chapter will be on the new methodologies that are developed to circumvent the limited fleet-coverage in the current observer programmes. The research focus on estimation methods for the discarded amounts is mainly focused on commercial (TAC) species, but they may also increase our ability to include data on non-TAC species (Dörner *et al.*, 2014). The two main identified methodologies are: (1) fishermen as empirical data source, and (2) electronic monitoring through Closed-Circuit Television (CCTVs). Both methodologies are described in detail in the following two sections, and put in perspective in the third section of this chapter.

3.2 Fishery-dependent data: fishermen's involvement and self-sampling

The knowledge of fishermen and the data coming from various fishing trips could be one of the major sources of information for fisheries management. The spatial and temporal extent, and the

coverage of gear types and practices that fishermen employ, may all together provide one of the clearest pictures of the state of the marine ecosystem. The amount of empirical data from scientific surveys cannot ever be as large as the information that comes from fisheries' operations. While the potential richness of information from fisheries is widely acknowledged (Dörner *et al.*, 2014; Graham *et al.*, 2011; Schwach *et al.*, 2007; Wilson *et al.*, 2006), fisheries' science has not found a simple and consistent way to incorporate the completeness of this vast source of information, or when it has, it is being contested (Hind, 2014; Linke & Jentoft, 2014; Pauly *et al.*, 2013; Silvano & Valbo-Jørgensen, 2008). The approach of the thesis, for instance, has not included any direct source of information exclusively from fishermen, i.e. without scientific observers. This thesis is, from that perspective, a classical example of the work conducted in fisheries' science, and ignores local (fishermen's) ecological knowledge (LEK). Hind (2014) collected >500 research outputs from the last century which do include fishermen's knowledge. The author reviewed the contributions and concludes that fisheries scientists rarely integrate LEK, or when they do, LEK is only partially (and generally marginally) integrated. The integration is mainly limited to logbook data or quantitative surveys. LEK may, however, provide an alternative and empirical information source to assess fish stocks and the marine ecosystem. It is currently unclear how this can be achieved (Hind, 2014). The main reason for the exclusion of fishermen's information is the requirement to meet conventional, scientific standards and the rigid way in which information is needed for quantitative frameworks (Dörner *et al.*, 2014). A proposal for a drastic change in the set-up of conventional fisheries' science is outside the scope of this thesis, but required if the criticism of Hind (2014) is accounted for. This thesis is limited to acknowledging and highlighting LEK as a potential but alternative way of moving towards a better knowledge base of the marine ecosystem.

The current rigid format of fisheries' science is however not devoid of the input from fishermen. Three examples of how fishermen contribute to the existing frameworks are: (1) their cooperation during scientific surveys (de Boois *et al.*, 2014); (2) their contributions to the developments of alternative gears (Depestele *et al.*, 2014a; Visserijnieuws, 2013) and (3) their input in detecting trends in the distribution of commercial species through structured questionnaires or personal diaries (Macdonald *et al.*, 2014a). A fourth example where fishermen may give direct input from their fisheries into scientific assessments is called 'self-sampling'. Self-sampling is a term which implies that fishermen themselves sample their catches without the presence of a 'scientific' observer aboard the fishing vessel. Self-sampling increases the involvement of fishermen in the data collection of discards, and includes them in the conventional analytical assessments, or at least in the first step, the data collection. The main driver is a significant increase in the number of data which are required for scientific assessments of fishing mortality (Searle *et al.*, 1999). This creates the possibility of

collecting a larger number of samples from many fishing trips rather than many hauls within a trip. Inter-trip variability of discards is generally higher than within-trip variability (Borges *et al.*, 2005; Rochet *et al.*, 2002), which renders the data collected by fishermen across several trips a valuable source of data in comparison to observer-based discard data.

The Norwegian fishermen and scientists demonstrate that collaboration may provide catch data for a number of commercial species. Catch data and biological samples are collected since 2001 from the so-called 'reference fleet' (Helle & Pennington, 2004). The reference fleet is composed of commercial fishing vessels (longliners, purse seiners) that are paid to measure a subsample of fish from selected catches and, less frequently, to take and preserve otolith, stomach, and genetic samples. The reference fleet demonstrates that fishermen can be included in the process of scientific assessments, although it also highlights the concern of Hind (2014). The established order of management advice from 'knowledgeable' scientists is not changed through the concept of a 'reference fleet'. Fishermen are merely included as scientific observers rather than scientific experts (Bjørkan, 2011), while scientists keep their focus on precision and other conventional, scientific ways of evaluating the provided data (Pennington & Helle, 2011). The Norwegian reference fleet shows, however, that valuable information can be retrieved from the collaboration with fishermen. The major drawback is the increasing cost to keep fishermen involved. In 2009, 25 scientists were working on the project with an approximate budget of 7.1 million USD (Bjørkan, 2011).

The common approach in other European countries cannot rely on an equal budget (Hoare *et al.*, 2011; Kraan *et al.*, 2013; Lordan *et al.*, 2011; Mangi *et al.*, 2014). The participation of fishermen is by consequence a serious matter of concern, as the merits of self-sampling are only warranted when participation is high, i.e. when fleet coverage is substantial. All studies indicated that incentives to cooperate were needed to increase the participation of fishermen to the self-sampling programmes. Financial compensation seemed to be an important driver for fishermen to join, either directly (being paid, Mangi *et al.*, 2014) or indirectly (increasing the possibility for being chartered as a vessel for fishery surveys) (Hoare *et al.*, 2011; Lordan *et al.*, 2011). When the expectations of the fishermen are not met, participation may suffer fatigue, stressing the difficulty to maintain this sort of programmes on a voluntary long-term basis.

However, when dedicated project management from scientists was involved, fishermen could deliver the data with the required scientific rigour to be included in analytical assessments. Achieving this objective required the provision of strict protocols and standardisation of the input from fishermen across participating vessels. The data were a compromise between the scientific objectives ('the more data of high quality, the higher the possibilities for a sound scientific analysis') and the

practicalities for the skipper ('fishing is making money in the first place'). Two main practices were involved: (1) training fishermen to collect catch data themselves and (2) obtaining a subsample from the catch which is subsequently analysed in the lab by fisheries scientists. The involvement of scientific observers as a liaison between the scientists in the lab and the fishermen at sea was found a good basis to communicate different expectations between both sides. Fishermen, for instance, may expect the output of their work quite rapidly and concrete, while scientists rather generalise the sources of information and obtain analytical results on a longer term. Communications between scientists and fishermen was crucial in order to match the divergent expectations from both sides.

The successes of the self-sampling programmes depended on the participation of fishermen. Data were best obtained when protocols were simple and easy-to-use. Several projects have validated these data with varying results. Some studies indicate that there is a high variability between scientists and fishermen's data: both under- and over-estimations of discards (Uhlmann *et al.*, 2014b) or under-estimation due to (infrequent) removal of the larger discarded individuals (Lordan *et al.*, 2011), while others stress that the new data may be robust and improve precision (Celic *et al.*, 2014; Hoare *et al.*, 2011; Mangi *et al.*, 2014). The Dutch self-sampling programme indicated differences in discard percentage between self-sampling (17%) and the observer programme (29%) with lower discard levels in self-sampling programmes. Catches of benthic organisms, sole and cod were in contrast lower in the observer programme. The differences were attributed to the low sampling coverage of observer programmes and the influence of one trip on the overall discard percentage (Rasenberg *et al.*, 2013). While each of these studies concludes that the reliability is high, there are also concerns raised on the validity of the data (without clear indication of its cause), which stress the utility to use third-party verification to improve data quality (Faunce *et al.*, 2015).

3.3 Fishery-dependent data: Fully-documented fisheries and remote electronic monitoring

Fully-documented fisheries (FDF) can be defined as a management system in which fishing mortality is calculated from total catch data instead of landings exclusively. A catch quota system is obviously more accurate than a landings' quota system, which is one of the reasons for the reform of the CFP. While catch quota have existed for a long time outside Europe, the concept of FDF became only recently a full-on topic in Europe. The introduction of Remote Electronic Monitoring (REM) is one of the eye-openers in that respect (Kindt-Larsen *et al.*, 2011). Catch quota, catch shares or Individual Transferable Quota systems generally prohibit discard practices of quota species. Discards of groundfish species for instance are not allowed in Nova Scotia (Canada), and New Zealand only

allows discards for species where discard survival is 'high', such as lobster (Sanchirico *et al.*, 2006). EU-oriented research is generally directed on recording the total catch of particular species, and does so through the combination of landings and discard data. REM systems were specifically designed to monitor both data sources at once, i.e. to monitor the complete catch. REM-systems may therefore be considered as a close link to FDF.

REM-systems collect data from a system which integrates video cameras, gear sensors and a geographical positioning system (GPS) to register all catches of particular species. Sensors aid in the identification of fishing activities by indicating the start of hauling or setting the gear (e.g. a photoelectric drum rotation sensor, a hydraulic pressure sensor) (Ulrich *et al.*, 2015). Camera images are recorded, stored on-board and transferred to the lab for further processing. Weight estimates of the catch are based on visual inspection of the fish lengths from CCTV images and subsequent length-weight conversions (van Helmond *et al.*, 2014) or may be provided by analysing the number of baskets of a standardized discard weight, which were held in front of the camera (Ulrich *et al.*, 2015). About 10% of the video images is selected for in a semi-random or stepwise selection procedure, and subsequently analysed in the lab. The selection of the video images was preceded by two steps in Van Helmond *et al.* (2014), i.e. (1) matching all trips that have video recording with logbooks of fishermen, and (2) evaluating image quality. Ulrich *et al.* (2015) included at least one haul from the last five hauls of a trip in the selected 10% images, because highgrading is expected to occur more frequently at the end of a fishing trip. The resulting weight estimates of the catch (discards and/or landings) are compared to the estimates from observers or the logbook estimates from fishermen. Fishermen also collected information on each of the individual fishing operations (e.g. setting and hauling times, locations, total catch weight, and so on.) and the weight of the discards for the total catch.

REM-studies generally indicate a good match (<30% error, Stanley *et al.*, 2011) between logbook and video observations when there are only few species caught and investigated (Ulrich *et al.*, 2015). There was a mismatch between logbook data from fishermen and REM-systems when total catch volumes are high and the investigated species is only caught in low volumes (van Helmond *et al.*, 2014). There are several plausible explanations, such as misidentification of species on camera views, errors in estimating lengths from the camera view or biases in length-weight conversions. Similar conclusions were drawn from tropical tuna fisheries, where the catch of the three target species (*Thunnus obesus*, *Katsuwonus pelamis* and *Thunnus albacores*) did not differ significantly between the REM-system and observer estimates (Ruiz *et al.*, 2014). In contrast, there was a mismatch between the number of recorded by-catch species in both methodologies. REM-systems were also tested for crab and lobster fisheries (*Cancer pagurus* and *Homarus gammarus*). The REM-systems

tended to slightly under-estimate the carapace width of crabs (mean difference of 0.853 \pm SD: 0.378 mm), while the mean difference was small for lobster (0.085 \pm 0.208 mm). Sex allocation was 100% accurate for crabs and lobsters >86mm, but decreased to 51% for lobsters <70mm. The error attributable to using video instead of manual measurements was <3mm, which is sufficient to detect growth increments (Hold *et al.*, 2015). The authors found camera systems to be a suitable method for collected data on the sizes and sex of crabs and lobsters. REM-systems are operational in various fisheries around the world (Mangi *et al.*, 2013): the British Columbia hook and line groundfish fishery targeting halibut (covering 200 vessels or 10,000 days at sea; Stanley *et al.*, 2014), the tropical tuna purse seine fishery, the Bering Sea flatfish trawl fishery, inshore set net and trawl fishery off the Canterbury coast in New Zealand and in the Australian gill net fishery. The applicability of REM-systems across the world indirectly indicates that these types of monitoring systems can be applied in real life. Indeed, REM-system may be an adequate tool, which is considerably more cost-effective than observers if good coverage is required (Ulrich *et al.*, 2015). Estimates of cost-effectiveness differ from a quarter of the daily cost of an observer-based system, over about one third to as low as one tenth of the cost of using observers (Needle *et al.*, 2014).

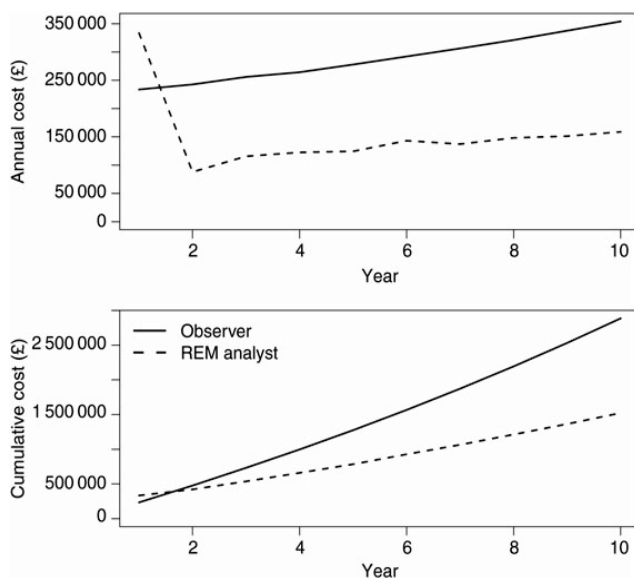


Figure 3.1 Summary of annual costs for simulation study of discard observer programmes using at-sea observers or Closed-Circuit Television of Remote Electronic Monitoring systems (REM) (after Needle *et al.*, 2014).

Needle *et al.* (2014) have simulated the costs of their sampling programme for the demersal whitefish fleet in 2012 (21 REM vessels, 4 observers and 2 REM analysts) (Figure 3.1). While the initial investments in REM systems were high, the annual costs decreased over time and rendered REM-systems more cost-effective than observers systems on the mid-and long-term. Although the calculations of cost-effectiveness are simplistic (e.g. they do not include validation by observers and REM-systems do not obtain data on biological samples such as age, sex, maturity, and so on), they do indicate the scope of REM-systems.

The REM-systems require an initial, capital investment as well as a certain ethical cost (cameras can be viewed as an invasion on fishermen's privacy) (Dörner *et al.*, 2014). The fleet coverage is likely to be higher, but the quality of the data is primarily directed to a limited number of species (<10 species) and a limited number of parameters (generally numbers by lengths, and/or sex for invertebrates) (Hold *et al.*, 2015; Ruiz *et al.*, 2014). REM-systems could, however, provide other benefits, such as the monitoring of accidental by-catch of charismatic species (Kindt-Larsen *et al.*, 2012) or spatial-temporal distribution of fishing effort in addition to VMS-distributions (Needle, 2015; Needle *et al.*, 2014).

3.4 Fishery-independent data: modelling discards

Currently, none of the methodologies have an established and applied way of sampling the total catch in terms of species, besides sizes with a certain (commercial) species. FDF generally accounts for the total catch of commercial fish species, but sets aside any clear data outcomes on the species composition of the catch. Several authors (Jennings, 2013; Le Quesne & Jennings, 2011; Thorpe *et al.*, 2014) put forward the Pope's postulate in this respect (Pope *et al.*, 2000):

'Fishing fleets generate a fishing mortality on non-target species which is less than or equal to that generated on the target species'

Shephard *et al.* (2015) have recently tested the postulate for 14 non-target species in the Celtic Sea. The biomass of the species was calculated from the Irish Groundfish Survey and the harvest rate was based on annual records of discards from the Irish discard observer programme. The harvest rates of cod and whiting that were calculated from the survey data and the discard observer programme were validated through a comparison with the cod and whiting estimates of fishing mortality from analytical assessments for these two stocks. Both estimates compared reasonably. The fishing mortality of at least one of the non-target species exceeded the mortalities of cod or whiting, challenging Pope's postulate. This study exemplifies that ignorance of non-target species in discard registrations may lead to mortality levels which reduce the population biomass to levels below acceptable level, such as Maximum Sustainable Yield (MSY) (Shephard *et al.*, 2015).

Heath & Cook (2015) also modelled the discards using data from scientific trawl surveys and focused on the North Sea from 1978 till 2010. A distinction between TAC and non-TAC species was made to put the results of their study into the perspective of the landing obligation. The percentage of non-target species in the catch is below 10% during the full length of the observation period (Figure 3.2). The overall trend in discarded biomass, however, indicates that the proportional importance of

discarded non-TAC species slightly increases over time. This is also true for the TAC-species: the contribution of discards to overall fishing mortality increases over time. The overall decline in total quantities of discards is thus mainly due to the reduction in the total catches rather than the reductions in discarded proportions. Heath & Cook (2015) also noted that the composition of the discards shifted from predominantly roundfish to >50% flatfish. Dab may be a typical example of a non-TAC and non-targeted species which is included in the discards. About one fourth of the discards comprise undersized plaice, which has been the major discarded species for the last two decades.

The discards of the Belgian beam trawl fishery are reported for plaice and dab in the North Sea (chapter 2), but did not include any estimates of discarded quantities of gurnards in any of the regional seas where Belgian beam trawlers were operating over the period 2009-2012 (Vanellander *et al.*, 2015). The discarded proportion of dab varied between 0.40 and 0.76 between 2009 and 2012. No data on the discarded proportion of gurnards were available, although the importance of gurnards was not negligible in both the North Sea and the Celtic Sea estimates (Rochet *et al.*, 2002).

The modelling exercises of Shephard *et al.* (2015) and Heath & Cook (2015) illustrate the potential of using scientific surveys to complement the fisheries dependent data sources, such as the observer programmes. These studies illustrate that the mortality experienced by non-commercial species may substantially alter their population structure, and that the biomass of these species may be among the top ten in a regional sea, indicating one aspect of their potential role in the ecosystem. The focus of both studies was on non-target fish species. Species of other ecosystem components than fish may also be significantly affected by being caught in fish nets. Benthic invertebrates for instance may constitute a large part of the discards in the two fisheries with the highest discard rates in the North Sea, i.e. beam trawl and *Nephrops* fisheries (Catchpole *et al.*, 2008a). The catch of marine mammals and birds is not considered as discards *sensu strictu*, but is defined as by-catch. The by-catch of these species may nevertheless compromise their populations levels of these ecosystem components as their survival chances are close to nil when being caught (Senko *et al.*, 2014; Zydalis *et al.*, 2013). Accuracy of catch data is also important for biodiversity implications (Bjelland & Wienerroither, 2014) and endangered species for which low levels of mortality may constitute a threat (Cressey, 2015; Sims & Simpson, 2015). The number of studies that include species without an economic interest is limited. The full suite of discarded species is sampled in several observer programmes but not in Belgium. Discards of non-target species have been occasionally recorded (Depestele *et al.*, 2012; Depestele *et al.*, 2008b; Van de Walle, 2012; Vandendriessche *et al.*, 2008).

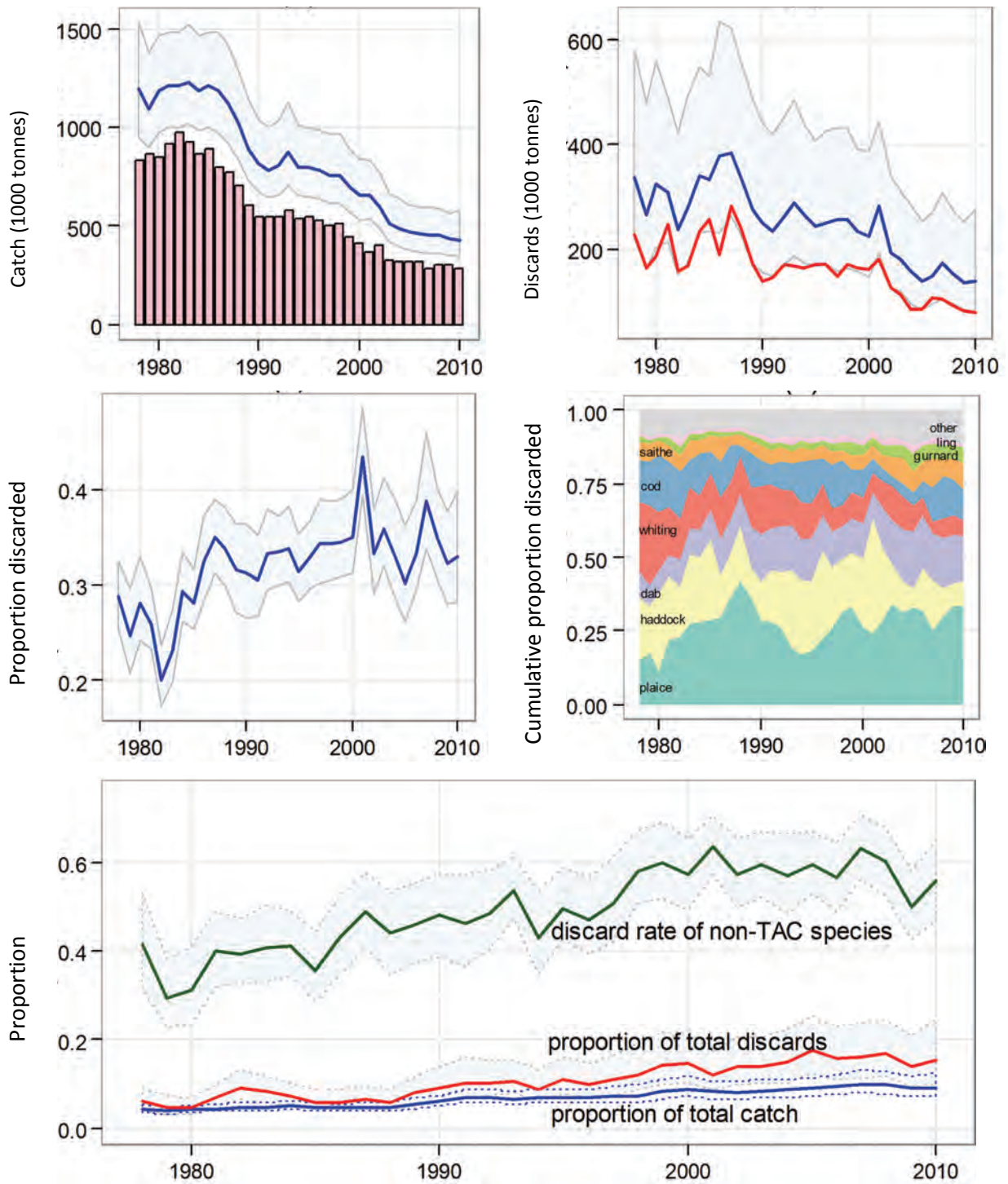


Figure 3.2 Model output from Heath & Cook (2015). Upper left panel: total annual catch (blue line: mean model estimates) and measured landings (bars); Upper right panel: modelled quantities of discards of demersal fish (blue line: mean, red line: weight of the fish discarded due to size); Middle left panel: mean annual discarded proportion of total demersal fish catch; Middle right panel: mean proportions of the eight most abundant species in the discarded weight of demersal fish; Lower panel: catch and discards of non-TAC species as proportions of the total demersal catch and discards, and the discard rate (= proportion of the catch discarded) for non-TAC species. Shaded areas indicated the 95% Credible Interval (Bayesian confidence interval).

4 Case study on modelling discards

Published as

Depestele, J., Vandemaele, S., Vanhee, W., Polet, H., Torrelee, E., Leirs, H., and Vincx, M. 2011. Quantifying causes of discard variability: an indispensable assistance to discard estimation and a paramount need for policy measures. – ICES Journal of Marine Science, 68: 1719–1725.

4.1 Abstract

Fishery-dependent data underpin the scientific advice given to fishery managers. However, discard estimates are often imprecise due to limited sampling coverage. Estimating discard rates from the length-frequency distributions (LFDs) in commercial catches may complement information from observer trips. Accurate estimates depend greatly on careful investigation of the discard variability. In this study, the impact of three essential factors was quantified for beam-trawl fisheries in the southern North Sea: (1) market prices, (2) landings-per-trip (LPT) limitations and (3) selectivity of the commercial fishing gear. Observed discard rates for cod, plaice, sole and whiting were compared with estimates based on the length-frequency data, taking into account the variability due to LPT limitations and market price. Observed discard estimates of cod and whiting differed significantly from LFD derived estimates due to highgrading. The results indicate that LFD derived discard estimates are only reliable if the crucial driving factors are quantified. LFDs can easily be collected from research vessels or by fishers in partnership with scientists. Based upon many of these LFDs and the discard-variability factors identified in observer programmes, discard rates can be better estimated.

Keywords: beam trawl, discards, fishery management, highgrading, market price, minimum landing size, North Sea, quota

4.2 Introduction

North Sea fisheries are responsible for the highest level of discards in the world (Kelleher, 2005), although discard rates have been reduced in recent years (Aarts and Poos, 2009). Fishery management is widely expected to reduce fisheries discards (Enever *et al.*, 2009) and especially discards of the beam-trawl flatfish fishery in the North Sea (Catchpole *et al.*, 2008). This fishery is among the greatest generators of discards in the North Sea (Catchpole *et al.*, 2005).

Reliable estimates of the discard rates and causes for their variability are of paramount importance both for stock assessments (Dickey-Collas *et al.*, 2007) and satisfactory fishery management (Rochet and Trenkel, 2005). Currently, estimates of discards are highly imprecise, owing to the limited coverage of the fleet for discard sampling (Dickey-Collas *et al.*, 2007).

Estimating discard rates from gear selectivity characteristics, as an addition to observer data, may provide valuable information for commercial fish species. Several exercises illustrate this theory, e.g. for discards of non-commercial fish species (Piet *et al.*, 2009), for historical discards (Aarts and Poos, 2009), and for studying spatio-temporal variation in discarding (Welch *et al.*, 2008). Discard rates can be inferred from the assumption that commercial fish below the minimum landing size (MLS) are discarded, whereas fish above MLS are categorized as landings. The discarding of marketable fish, defined as highgrading (Gillis *et al.*, 1995a), can reduce the accuracy of such discard estimates (e.g. Piet *et al.*, 2009). Length-frequency distributions (LFDs) for each type of commercial fishing gear could be obtained from gear-technology trials or directly from fishers' measurements. Additionally, they can be deduced from selectivity parameters and estimates of absolute abundance. The latter are obtained by raising fish-density data while taking account of the catchability of the survey gear (see Piet *et al.*, 2009). Adding this method of discard estimation to the present monitoring programmes could expand the number of records and has the potential to extend the spatio-temporal distribution of discard data at low cost (Polet *et al.*, 2010).

Discard estimates from modelling exercises depend on fish abundance, fishing effort, MLS, and gear selectivity (Piet *et al.*, 2009). Market incentives and policy measures have been identified as crucial drivers of discarding. There is generally no quantitative assessment of their contribution to the discarding problem (Rochet and Trenkel, 2005).

This paper elucidates the market- and policy-driven causes of discarding. To illustrate this, four commercial fish species were selected from the Belgian flatfish beam-trawl fishery in the southern North Sea, namely Dover sole (*Solea solea*), plaice (*Pleuronectes platessa*), cod (*Gadus morhua*), and whiting (*Merlangius merlangus*). Understanding the reasons for discarding may have implications for

policy measures such as quota restrictions and gear regulations. It also clarifies what parameters other than gear selectivity and MLS should be taken into account when estimating discards of commercial fish species.

4.3 Materials and methods

4.3.1 *The Belgian discard-sampling programme*

This study focused exclusively on the Belgian flatfish-directed beam-trawl fishery in the southern North Sea (ICES Division IVc) during 2006, 2007 and 2008. Fishery-dependent data have been collected for the Belgian beam-trawl fleet in accordance with the EC Data Collection Regulation (EC, 2000). These included landing and discard data from commercial vessels, obtained since 2004 via on-board observation. Selection of the vessels for this observer programme was random but conditional on the cooperation of the fishers. The data were collected during fishing activities in one or more ICES Divisions, e.g. IVb and IVc (Figure 4.1). Fishing trips were defined as the primary sampling units, hauls within a fishing trip as secondary sampling units. Hauls are correlated within trips. This implies that hauls are not independent and could confound statistical analyses (see further below). The sampled gear was a beam trawl, 4 or 12 m wide, equipped with a chain mat and an 80 mm codend.

The sampling coverage ranged between 2.4% and 4.5% of landings, which is comparable to other discard-sampling programmes (e.g., Stratoudakis *et al.*, 1998; Rochet *et al.*, 2002). The spatial distribution of the sampling is a good reflection of the commercial fishing activity in the southern North Sea (Figure 4.1), which was primarily in the western part of that area⁵. A total of 28 trips (374 tows) were sampled onboard nine different beam trawlers. Every alternate haul was sampled by the observers, unless that haul was rejected (e.g., due to severe net damage, <3%) in which case the next haul was sampled.

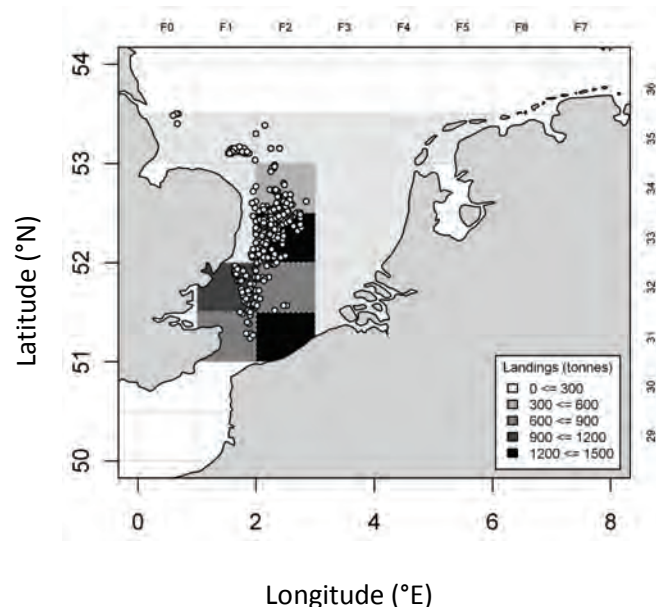


Figure 4.1 Spatial distribution of the total landings of sole, plaice, cod, and whiting by Belgian flatfish beam trawlers in the southern North Sea (shaded areas) and sampled hauls⁵ (dots).

⁵ Sampling did not cover the BPNS (chapter 2.4.2). Meta-data were sourced from D1-fishery-biodatabase (SQL).

Sampling took place around the clock to reflect typical working conditions. Fishers sorted the marketable fish from the conveyor belt, while the observers also collected all discarded commercial fish species, both undersized and highgraded. The observers did not interfere with fishing operations. However, an observer effect, resulting from changes in fishing practice and location, could potentially bias the observed discard rates (Liggins *et al.*, 1997; Benoît & Allard, 2009).

4.3.2 Discard rates and highgrading

The marketable and discarded fish samples were weighed and fish lengths were measured to the nearest centimetre below. In 68% of the hauls, the lengths of all individual fish were measured. Only when an observer judged that the abundance of a species was excessive, a subsample was taken of at least one 40-litre basket. To ensure that sufficient fish of each size category were sampled, observers identified the most obvious length modes (Cotter and Pilling, 2007). The total number of discards by length of a species in the relevant haul was estimated using the subsampled fraction and the observed length distribution. The mean of the raising factors⁶ was 1.54, with a minimum of 1.0 and a maximum of 10.8. LFDs were determined for sole, plaice, cod, and whiting. Sole and plaice comprised most of the landings, about 20% each, in contrast to cod and whiting which were considered bycatch species (7 and about 1%). In this context, bycatch is defined as commercial species which are not targeted but nevertheless occur in the landings. Discard rates, both in number and weight, were estimated directly from at-sea observations (D_{obs}). The discard rates, in number and weight, were also calculated from LFDs of the catch and MLS (D_{estim}). The discard rate (D) was defined according to Rochet and Trenkel (2005):

$$D (\%) = (d / (d + k)) * 100 \quad [1]$$

where d is the weight or numbers of the discarded fish (caught but not kept) and k is the ungutted weight or numbers of the harvested fish (caught and retained). D_{obs} therefore includes possible highgrading, because the discarded (d) and harvested (k) fractions are dependent on the sorting behaviour of the fishers. The ungutted weight of harvested fish, needed to calculate D_{obs} , was obtained using gutting factors of 1.04 for sole, 1.05 for plaice, 1.18 for whiting, and 1.17 for cod (EC, 2009b).

⁶ If a LFD was hypothetically estimated from the observation of 1000 fish, the total number of discarded fish was obtained by multiplication with the raising factor. A raising factor of 1.54 implies that the total number of discarded individuals was 1540 with a LFD following the pattern as observed in 1000 fish.

Calculating discard rates on the basis of LFDs and MLS (D_{estim}) partly reflects the selectivity of the beam trawl and does not account for the sorting behaviour of fishers. Fish below MLS were considered discards (d), whereas fish above MLS were assumed harvested (k). Consequently, highgrading was not considered in D_{estim} . Equation (1) gives D_{estim} in numbers. A length-weight conversion is needed to obtain the corresponding weight. Relevant data were collected in the southern North Sea during August and September of the years 2007–2008, during research surveys with the RV 'Belgica'. The total length of the fish (L , cm) was measured to 0.5 cm below, and ungutted wet weights (wt , g) were also recorded at sea. Numbers were converted to length using a weight-length relationship (Table 4.1).

4.3.3 *Factors affecting the highgrading of cod and whiting*

4.3.3.1 *Landings-Per-Trip, market price and catch composition*

Discard rates estimated directly from LFDs were compared to on-board observations using t-tests. Any discrepancies between observed and estimated discard rates may be related to market price, Landings-Per-Trip (LPT) limitations, and/or catch composition. These factors were evaluated as explanatory factors for discarding above MLS. Mean fish prices were retrieved from the fishing ports of Ostende, Zeebrugge, and Nieuwpoort. The LPT limitations for each sampled vessel were calculated from national regulations which specify the maximum allowable landings per trip-day on the basis of engine power. The maximum allowable landings varied over time, according to national management decisions, ensuring that the total landings of the Belgian fleet matched national quota restrictions. The maximum allowable landings of whiting were 250 kg per day for Belgian beam trawlers in 2006, whereas no limitations were set in 2007 and 2008. On 27 November 2008, the fishery for whiting was closed. Constraints on catch composition are set in EU regulations. The catch must consist of at least 70% of listed species at all times prior to return to port (EC, 1998). The list includes a wide range of species, including for instance sole, plaice, dab, brill and turbot. Cod is not in this list, but whiting is. To investigate the influence of catch composition, whiting was considered at risk of being discarded if the composition of trip landings had more than 75% of listed species. Similarly, cod could not exceed 20% of the landings. This factor was included in the analysis by considering cod discards to be likely if the trip landings for cod were over 15%.

4.3.3.2 Statistical analysis

A response variable⁷ R was created by the following equation to model the discrepancy for discard rates by weight:

$$R = D_{obs} / (D_{obs} + D_{estim}) \quad [2]$$

where D_{obs} is the observed discard rate and D_{estim} the discard rate estimated from LFDs and MLS. LPT limitations, fish price and the catch composition were *a priori* selected to explain this discrepancy. For whiting, the effect on the response variable was tested using a multivariate generalised linear mixed effect model (GLMM) with Gaussian distribution. Fixed effects were fish price, the nominal LPT limitations, their interaction, and the risk of exceeding the landings composition with listed species over 75% as another factor⁸. Fishing trips was included as a random, nesting factor to account for inter-haul correlations. Model selection was carried out using stepwise backward selection with the Akaike information criterium (AIC) as selection criterion. A similar approach was followed for modelling the highgrading of cod, i.e. before the MLS was shifted from 40 to 50 cm by national legislation (M.B. 14/12/2007). A GLMM was fitted to account for the random variability induced by fishing trip, and LPT limitations was included as a continuous variable. The homoscedasticity and normality assumptions were verified through a visual analysis of the residuals (not shown). However, the response variable could not theoretically have values above 1 and below 0.5 as D_{obs} cannot take values smaller than D_{estim} . Therefore a generalised additive mixed model (GAMM) was fitted. A two-dimensional tensor-product of cubic regression splines for LPT limitations and fish price was used in the full model (Wood, 2006). The final model of the backward selection was refitted with one-dimensional cubic regression splines and restricted maximum likelihood (REML) (Zuur *et al.*, 2009).

⁷ Observed (D_{obs}) and estimated (D_{estim}) discard rates are equal when R is 0.5. A response variable $R < 0.5$ indicates that the observed discard rates are lower than the estimated discard rates. The observed discard rates are higher than the estimated discard rates when the response variable $R > 0.5$, i.e. when high-grading occurs.

⁸ The risk of exceeding was quantified as a categorical variable: 'Yes' or 'No'.

4.4 Results

4.4.1 Discard rates and highgrading

To estimate the discard rates in weight, numbers-at-length were converted to weight-at-length, using length-weight relationships. Parameters a and b of the length-weight relationships are given in Table 4.1, together with the regression coefficient (Pearson r^2), the number of individuals measured (n) and the sizes of the smallest and largest individuals measured.

Table 4.1 Parameters of the length-weight relationship for sole, plaice, cod, and whiting in the southern North Sea during August and September of the years 2007–2008. The parameters a and b are estimated by $W = a \cdot L^b$ where W equals the observed wet weight (grammes) and L the total length (cm).

	a	b	Pearson r^2	n	Min (cm)	Max (cm)
Sole	0.007568	3.0617	0.9862	2724	5.5	40.5
Plaice	0.009641	3.0319	0.9935	2897	8.5	57
Cod	0.010137	2.9912	0.9904	83	15.5	62
Whiting	0.009030	2.9508	0.9898	1021	5.5	39

Discard rates are presented as numbers and weights for sole, plaice, cod, and whiting (Table 4.2). The differences between observed and estimated discard rates were significantly different ($p < 0.01$ in paired t-test) for all four species. Examining LFDs illustrates that the discard rates were apparently dictated by MLS for sole and plaice but less so for whiting and cod for which substantial discarding occurred above MLS (Figure 4.2). This is equally demonstrated by the higher standard deviations of the differences between observed and estimated discard rates for cod and to some degree for whiting. The LFD-curves for cod and whiting imply that factors other than MLS determine discarding. The mean proportion of discarded fish above MLS was calculated for each quarter in each sampling year (Table 4.3). Few cod were apparently highgraded in 2006, but this did occur in the last quarter of 2007 and in the first and second quarters of 2008. Whiting was highgraded in each quarter in each year, although highgraded proportions were lower in the second quarter of 2006 and the first of 2007. Highgraded proportions of whiting were higher in the fourth quarter of 2006 and the second quarter of 2008; no data were available for the first quarter of 2007. These findings suggested that discards of fish above MLS did not change predictably between years. Factors driving highgrading of cod and whiting were then investigated in more detail.

Table 4.2 Mean discard rates with standard deviation (s.d.) for sole, plaice, cod, and whiting in Belgian beam-trawl fisheries in the southern North Sea during 2006-2008. Upper rows: number-based discard rates; lower rows: weight-based rates. The observed rates have been determined at sea, whereas the estimates are obtained from length frequency distributions of landings and minimum landing sizes. Their differences indicate the validity of using LFD and MLS for discard estimation.

Number-based	Observed discard rate	Estimated discard rate	Difference observed and estimated discard rate
	Mean (s.d.)	Mean (s.d.)	Mean (s.d.)
Sole	0.29 (0.18)	0.25 (0.16)	0.04 (0.04)
	0.13 (0.11)	0.11 (0.10)	0.02 (0.03)
Plaice	0.42 (0.24)	0.39 (0.22)	0.02 (0.07)
	0.27 (0.21)	0.25 (0.18)	0.03 (0.10)
Cod	0.65 (0.29)	0.55 (0.31)	0.10 (0.27)
	0.47 (0.31)	0.36 (0.31)	0.12 (0.24)
Whiting	0.70 (0.33)	0.55 (0.29)	0.15 (0.51)
	0.61 (0.33)	0.46 (0.27)	0.15 (0.16)

Table 4.3 Mean proportion (standard deviation) of cod and whiting above the MLS discarded in the Belgian beam-trawl fishery in the southern North Sea by quarter (Q) in the period 2006 – 2008. Upper rows: number-based highgraded proportion; lower rows: weight-based proportion.

Highgraded proportion of cod discards				
	Q1	Q2	Q3	Q4
2006	0.03 (0.09)	0 (0.01)	0	0.04 (0.08)
	0.04 (0.09)	0.01 (0.02)	0	0.04 (0.09)
2007	No data	0.01 (0.05)	0.05 (0.11)	0.47 (0.35)
		0.01 (0.04)	0.06 (0.11)	0.37 (0.32)
2008	0.62 (0.25)	0.28 (0.26)	0.02 (0.06)	0.03 (0.13)
	0.49 (0.22)	0.31 (0.24)	0.04 (0.09)	0.03 (0.14)
Highgraded proportion of whiting discards				
	Q1	Q2	Q3	Q4
2006	0.17 (0.24)	0.07 (0.07)	0.18 (0.13)	0.23 (0.18)
	0.18 (0.23)	0.08 (0.04)	0.22 (0.08)	0.22 (0.14)
2007	No data	0.12 (0.13)	0.15 (0.09)	0.16 (0.11)
		0.12 (0.11)	0.23 (0.06)	0.20 (0.05)
2008	0.10 (0.12)	0.32 (0.24)	0.18 (0.08)	0.17 (0.13)
	0.15 (0.11)	0.37 (0.22)	0.26 (0.09)	0.23 (0.16)

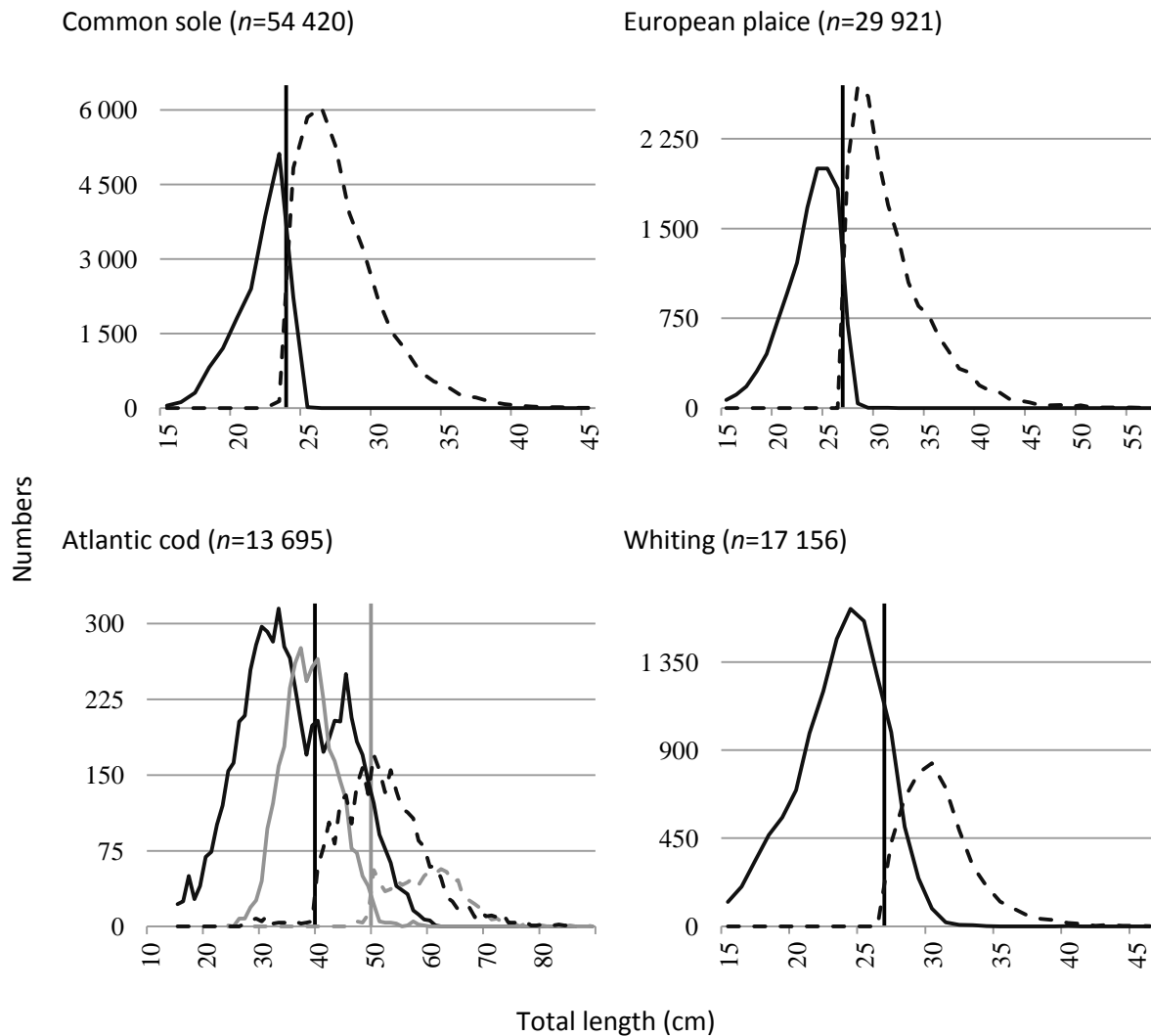


Figure 4.2 Smoothed length-frequency distributions of sole, plaice, cod, and whiting for the discarded (solid) and the landed fraction (dashed) based on on-board observations of the Belgian beam-trawl fishery in the southern North Sea in 2006 – 2008. The Minimum Landing Size (MLS) is indicated by a vertical line (24 cm for sole, 27 cm for plaice and whiting). The MLS of cod changed from 40 to 50 cm on 1 July 2008. Length-frequency distributions are shown before (black) and after (grey) this change. n is the number of individuals measured.

4.4.2 Factors affecting the highgrading of cod and whiting

4.4.2.1 LPT limitations, market price, and catch composition

The LPT limitations for cod varied over the years, without any obvious, repeated pattern (Figure 4.3). Fish prices were also variable: 2.9 to 4.5 euro per kg for cod and 1.0 to 2.4 euro per kg for whiting (Figure 4.3). Five fishing trips (120 hauls) gave landings of at least 15% cod, and on three fishing trips (58 hauls) the landings comprised less than 75% of listed species.

4.4.2.2 Statistical analysis

None of the examined explanatory variables in the GLMM could explain the variation in the response variable ($D_{obs} / (D_{obs} + D_{estim})$) for whiting, indicating that discarding of whiting above MLS was not due variation in fish price, LPT limitations, or to the catch composition. Backward selection of the fixed-effect variables for cod resulted in a model with LPT limitations as the only significant variable ($F = 6.806$, d.f. = 1.486, $p < 0.001$). The smoother had 1.486 effective degrees of freedom⁹, indicating a nearly linear decreasing trend of the response as a function of LPT limitations (Figure 4.4). However, the adjusted r^2 is 0.28, indicating that the explanatory power of this model was limited.

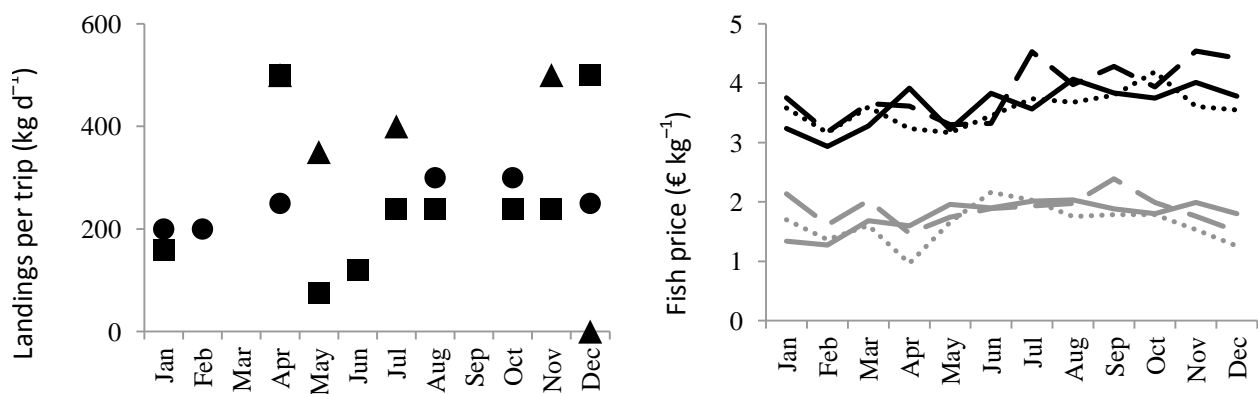


Figure 4.3 Cod Landings-Per-Trip are represented by month for each sampled trip in 2006 (square), 2007 (triangle) and 2008 (circle) (left). Variation of fish prices by month for cod (black) and whiting (grey) in 2006 (solid line), 2007 (dashed line) and 2008 (dotted lines) (right). (Source: Belgian Fisheries Service).

⁹ The amount of smoothing is expressed as effective degrees of freedom (edf). The higher the edf, the more non-linear is the smoother. A high value (8–10 or higher) means that the curve is highly non-linear, whereas a smoother with 1 degree of freedom is a straight line. (Zuur *et al.*, 2009)

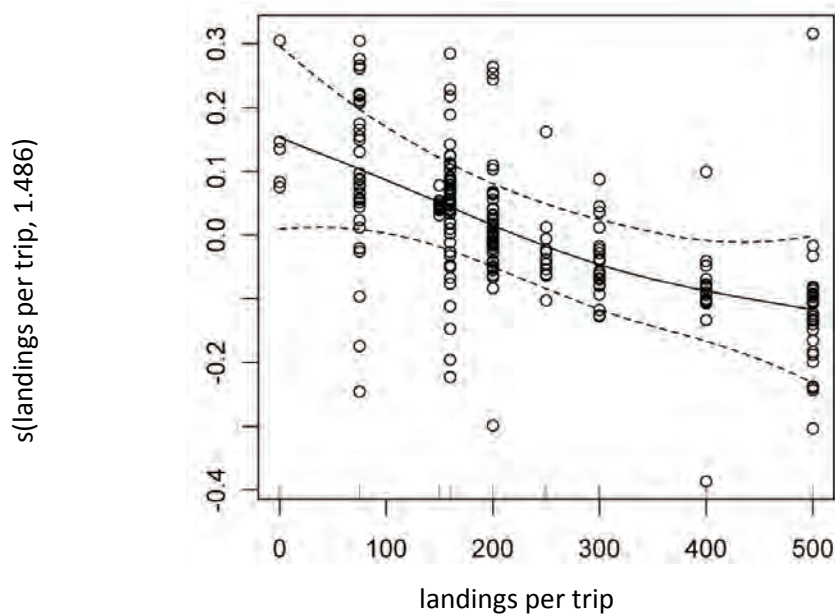


Figure 4.4 Non-linear smooth relationship between LPT limitations and the response variable in the final GAMM model for cod. The dotted lines are the 95% Confidence Interval

4.5 Discussion

This chapter investigates the discarding behaviour of the Belgian beam-trawl fishery in the southern North Sea. Sole and plaice were potentially highgraded. However, the low quantities indicate that highgrading is unimportant. Cod above MLS is possibly discarded due to LPT limitations, whereas discards of whiting above MLS cannot be attributed to seasonal changes in fish price, LPT limitations, or catch composition. Estimating discard rates of sole and plaice from LFDs and MLS is therefore recommended for the Belgian beam-trawl fishery in the southern North Sea. Additional factors need to be considered if discard rates are to be estimated for cod and whiting.

4.5.1 Highgrading and discard estimates of target species

The observed and estimated discard rates for all species were significantly different, though only a limited amount of sole and plaice above MLS was discarded, and also that few undersized sole and plaice were landed. These differences are more likely due to the visual sorting process. For example, fishers may unintentionally discard 27-cm plaice and retain undersized individuals. The limited landings of undersized fish and discards of fish above MLS contributed to the differences between the observed and estimated discard rates. Therefore, it is concluded that highgrading of sole and plaice is negligible in the Belgian beam-trawl fishery in the southern North Sea.

Kell and Bromley (2004) hypothesised that differences in highgrading can be largely explained by the interplay of targeting behaviour and LPT limitations. Targeting behaviour depends on species

abundance and fish price (Gillis *et al.*, 2008), and is hence indicated by the total landed value. The proportion of landed value was 49% – 53% for sole; 11% – 13% for plaice; 4% – 5% for cod, and < 1% for whiting for the Belgian fishing fleet in 2006 – 2008. Assuming sole and plaice are “true” target species for the Belgian beam trawlers in the southern North Sea, this study might indicate that fishers can eliminate highgrading of target species. For example, the choice of fishing grounds may avoid highgrading, as relative abundances of target species can be matched with LPT limitations (Gillis *et al.*, 2008; Quirijns *et al.*, 2008). If an excessive amount of marketable plaice is caught, fishers may redirect fishing effort towards patches with lower abundance. Bromley (2000) has similarly suggested that if marketable plaice is of low value (as is typical in early spring), fishing effort is reduced.

In contrast with these findings, highgrading of plaice has been suggested when the largest size-classes are in low condition. Highgrading low-value fish, typically done in early spring, is meant to save LPT for high-value fish at the end of the year (Poos *et al.*, 2010). The monthly plaice landings of the Belgian beam-trawl fishery ranged between 2% and 7% of the annual total from February to August, and 10%–18% from September to December. Combining these results with Figure 4.2 seems to confirm that Belgian fishers do not highgrade plaice in the southern North Sea. Interestingly, this contrasts results from the Dutch beam-trawl fishery. A possible explanation lies in the overlap between the spatial distribution of plaice and fishing effort. Landings of the Belgian beam-trawl fishery in the southern North Sea primarily originate from fishing grounds with a rocky seabed along the western part of ICES Division IVc (Figure 4.1). The Dutch beam trawlers mostly operate along the southeastern and central part of the North Sea (Poos and Rijnsdorp, 2007), where the abundance of plaice is higher (Bogaards *et al.*, 2009). However, avoiding LPT limitations on plaice could also be realised by misreporting plaice catches to a bordering ICES Division with unfulfilled LPT limitations (B. Deputter, pers. comm.). Finally, the differences in discarding behaviour could be due to an observer effect, which has been demonstrated in the Gulf of St. Lawrence (Benoît & Allard, 2009). Perhaps Belgian fishers chose not to highgrade sole and plaice because of the observers’ presence. A post hoc analysis of this effect is difficult to address quantitatively within the current design of the discard-sampling programme (Benoît & Allard, 2009) and hence is not tested in this study. However, the potential observer effect implies the need for caution when drawing conclusions on the apparent absence of highgrading of sole and plaice in the beam-trawl fishery. Understanding the preconditions of the fishery is advisable if sole and plaice discards are estimated from LFD and MLS.

4.5.2 *Highgrading and discard estimates of bycatch species*

Discard rates of bycatch species must account for market- and management-induced variability. Species with low abundance and value lack profitability and are therefore more likely to be highgraded. This was formally tested for whiting, which was highgraded in nearly all quarters (Table 4.3) independent of LPT limitations. Stratoudakis *et al.* (1998) obtained similar findings for demersal trawlers and seiners catching low-value gadoids. For bycatch species with a low profitability, LPT limitations were not the main limitation. Vessel-storage capacity and/or catch composition should reveal the incentives for highgrading, especially when low-value species are discarded to make space for more profitable fish (Gillis *et al.*, 1995b), when catches of target species are below expectations (Redant and Polet, 1994), or when legislation requires the bulk of the landings to be composed of the target species (e.g. Catchpole *et al.*, 2008; Benoît & Allard, 2008). The analysis of highgraded whiting in this study did not indicate catch composition at the trip level as an important driver. However, examining the high variability of catch composition on a haul-by-haul basis might uncover a relationship between catch composition and highgrading of bycatch species. Our analysis shows substantial cod discarding above MLS during periods with stringent LPT limitations. Catch-composition requirements might also have influenced highgrading, but a haul-by-haul approach for detecting this effect is again preferable. Owing to the lack of landings and discards data by haul, the hypothesis could not be tested.

4.5.3 *Management implications*

This study has indirect implications for fishery management. First, it indicates that estimation of discard rates may complement the data available from discard-observer programmes. This could lead to the improved stock assessments and advice on total allowable catches. A protocol for such estimates should require that commercial species are regulated by MLSs, and that LFDs of the catch using a particular fishing gear can be estimated for the species under investigation. One conceivable way of collecting those LFDs would be to involve fishers in combination with electronic monitoring and automated observations of fish species and lengths using a digital camera (White *et al.*, 2006; Benoît & Allard, 2009). Another way to obtain LFDs of fish catches using commercial gear would be through the combination of abundance estimates and selectivity parameters (Piet *et al.*, 2009). As LFDs for each type of fishing gear and fish species could be more easily collected than actual discard data, their spatial and temporal range could be expanded considerably. However, estimates would need to be verified against the observed discard rates from observer programmes, as variability of discarding can also result from market- and management-induced factors (Rochet and Trenkel, 2005). Such a protocol, when applied to the Belgian beam-trawl fishery in the southern North Sea,

would result in justifiable estimates for the target species (sole and plaice). For highly profitable bycatch species (e.g. cod), LPT limitations need to be accounted for when making an estimate. For bycatch species with low profitability (e.g. whiting), LPT limitations and seasonal variations in fish price cannot explain the variation in discarding. For low-value species, a more detailed investigation of discard-variability factors is needed, such as fishers' behaviour in response to catch composition.

Secondly, specifying the drivers of discard variability expedites the identification of discard-mitigation measures and the evaluation of introduced actions (Borges *et al.*, 2006; Enever *et al.*, 2009). Whiting is a clear example where gear selectivity needs to be changed in order to avoid the conditions leading to highgrading. Highgrading occurs in nearly all quarters, although there are feasible techniques that reduce catches of whiting without major losses of sole and plaice (van Marlen, 2003). In addition to establishing factors of discard variability, and mitigating the causes, these results must be followed up. Only in this way will fishery management succeed (Graham *et al.*, 2007; Enever *et al.*, 2009).

4.6 Acknowledgments

The authors wish to thank the ship owners, crew and on-board observers for their assistance in the ILVO discard sampling programme. This research was facilitated by information from the Administrative Centre of the Department for Agriculture and Fisheries of the Flemish Government (Eddy Tessens). We are also greatly indebted to Professor Stefan Van Dongen for helpful discussions and valuable suggestions on the statistical analyses. We thank Miriam Levenson for the valuable linguistic comments. The authors are grateful for advice of the IJMS guest and in-house editor and two anonymous referees, as this advice considerably improved the quality of this paper. This study was partially carried out within the WAKO-II project, funded by Contract SD/NS/08A of the Belgian Science Policy. Further support for this work was provided by the Flemish Government and the European Commission (Data Collection Framework).



*Short-term
survival
of discards*

PARTIM II

Discards from human and stock perspective

5 Short-term discard survival

Published as

Depestele, J., Desender, M., Benoît, H.P., Polet, H., Vincx, M., 2014. Short-term survival of discarded target fish and non-target invertebrate species in the "eurocutter" beam trawl fishery of the southern North Sea. *Fisheries Research*, 154: 82-92.

5.1 Abstract

Few studies have examined discard survival in beam trawl fisheries, especially in 4 m beam trawl fisheries using chain mats and limited haul durations. This so-called "eurocutter" fishery is carried out by beam trawlers with an engine power ≤ 221 kW and is allowed in the 6 to 12 nm zone in contrast to larger beam trawlers which operate solely outside of the 12 nm limit. Chain mat beam trawling was developed to prevent large boulders from entering the net, and is typically conducted at lower fishing speed than tickler chain beam trawling. This study obtained short-term survival estimates for this "eurocutter" fishery by monitoring post-capture mortality in tank-held organisms. Survival was high to very high ($>75\%$) for benthic invertebrates, but not for fish. All examined whiting (*Merlangius merlangus*) and pouting (*Trisopterus* sp.) died. Only 14 % of sole (*Solea solea*) survived to 91 h of observation, and 48 % of plaice (*Pleuronectes platessa*) to 77 h. The survival probability was higher for cod (*Gadus morhua*) (66 % to 88 h) and skates (*Rajidae*) (72 % to 80 h). However, the mortality rate had not stabilized within the period of observation. Survival models were used to estimate the minimum duration of captivity required to properly evaluate short-term survival, and to investigate the role of physical injuries and other pertinent covariates (catch weight, fish length, fishing depth, salinity, sea surface temperature, air temperature and fishing trip) in determining fish discard survival. The results of this study indicate a high variability in discard survival amongst taxa and highlight that physical injuries when taken alone are a limited proxy for survival of 4 m beam trawl discards and that small fish specimens have a limited chance of surviving discarding.

Keywords: beam trawl, discard mortality, physical injuries, survival proxy, survival analysis

5.2 Introduction

The overall ecological impact of beam trawl fisheries is amongst the highest of different gear types (Suuronen *et al.*, 2012). In addition to the well-documented effects on benthic habitats, discarding is also of particular concern. The UK beam trawler fleet discards approximately one third of the weight of their fish catch in the North Sea or two thirds in numbers (Enever *et al.*, 2009), while the German flatfish-directed beam trawler fleet discards between 56 and 72 % of their total catch weight, i.e. including non-commercial fish and benthic invertebrate species (Ulleweit *et al.*, 2010). Discard rates of individual fish species in numbers in UK fisheries (beam trawls, otter trawls and *Nephrops* otter trawls) vary between 15-20 % and >75% (dab and gurnards). Discard rates of individual fish species in weight in the German beam trawler fleet also vary between low values (2-8% for sole, brill, turbot) and > 90% (whiting, gurnards). Despite a number of initiatives to reduce discards in beam trawl and other fisheries, the European Commission (EC) has deemed progress to be insufficient and has therefore proposed a ban on discards of commercial species (EC, 2011). However, decision making concerning a ban is on-going and survival of fishery discards is a ponderous subject of debate (EC, 2013). Information is required on the relative conservation benefits that might arise from accounting for all fishery catches as part of a ban in which all discards die, versus those arising from regulations that allow for discarding of certain species, with ensuing survival of some organisms.

In practice, considerable efforts are made to understand discard amounts, but relatively little is known about the survival of discarded organisms. Formal estimates of discard survival are difficult to obtain due to the complex logistics for survival studies (see review in Broadhurst *et al.*, 2006). A number of those survival studies of discards in beam trawl fisheries were conducted mainly in the early 1990s. They focussed primarily on beam trawling with tickler chains and either very short (≤ 0.5 h) or long hauls (≥ 2 h) (Table A5.1). This study focuses on the “eurocutter” fishery with 4 m beam trawls and chain mats and with haul durations of approximately 1.5 h. Beam trawling with tickler chains is typically conducted at higher fishing speeds than with chain mats (Rijnsdorp *et al.*, 2008). Also, in contrast to tickler chain beam trawling, chain mat trawling can be conducted in rocky fishing grounds as the chain configuration prevents boulders from entering and tearing up the net. Given that haul duration (Van Beek *et al.*, 1990), catch composition and towing speed affect fishing induced stress, injuries and survival (Davis, 2002), differences in survival between trawls with tickler chains and chain mats are expected (e.g. Lindeboom and de Groot, 1998: 170). “Eurocutter” beam trawlers have an engine power ≤ 221 kW and have different fishing rights than larger vessels. They are allowed to fish in the 6 to 12 nm zone and in the plaice box (Beare *et al.*, 2013), thus exhibiting different fishing patterns than larger vessels (Poos and Rijnsdorp, 2007). Differences in the environment in which fishing takes place (e.g. depth, salinity, temperature) may also influence

discard survival. The Dutch and Belgian ‘eurocutter’ fishery mainly takes place in the southern North Sea (Taal *et al.*, 2010; Tessens and Velghe, 2010; Van Hal *et al.*, 2010). Dutch ‘eurocutters’ predominantly fish with tickler chain beam trawls during the summer period in the southeastern North Sea, whereas most of the Belgian beam trawl landings and discards originate from the winter period (Marchal, 2006; Tessens and Velghe, 2010; Chapter 4). Although the ‘eurocutter’ fleet is small (10.7 % and 19.6 % of the Dutch resp. Belgian beam trawler fleet in 2009), the envisaged differences in discard survival between beam trawl fisheries could lead to different advice for the “small” and “large” beam trawler fleet in the framework of the discard ban, thereby motivating this study.

This study had three main objectives. The primary aim was to obtain estimates of the short-term survival of a wide range of discarded organisms in the “eurocutter” fishery. The fish species selected in this study were those that constituted most of the discards in the fishery and represent a diversity of biological characteristics (Chapter 4; Silva *et al.*, 2012; Uhlmann *et al.*, 2011): two flatfish species, i.e. sole (*Solea solea*) and European plaice (*Pleuronectes platessa*), three roundfish species, i.e. whiting (*Merlangius merlangus*), pouting (*Trisopterus* sp., >90 % *T. luscus*) and cod (*Gadus morhua*), and skates (Rajidae) for the elasmobranchs. The survival of benthic invertebrate species was also examined to investigate the effect of longer haul durations on survival, as the only “chain mat” study on their discard survival was conducted during 30 min hauls (Kaiser and Spencer, 1995). The selected invertebrates were common starfish (*Asterias rubens*), ophiurids (*Ophiura* sp.), edible crab (*Cancer pagurus*), hermit crab (*Pagurus bernhardus*), sea mouse (*Aphrodita aculeata*), green sea urchins (*Psammechinus miliaris*) and swimming crabs (*Liocarcinus* sp., of which >90 % were *L. holsatus*).

The second aim of the study was to evaluate whether the degree of injury sustained by an organism can predict eventual discard survival in the “eurocutter” fishery. The relationship between injuries and discard survival has been found for invertebrates and fish in other fisheries (e.g. Enever *et al.*, 2008; Benoît *et al.*, 2010; 2012). The benefit of defining such relevant proxies for discard survival is that they represent a much more cost-effective manner of evaluating and that they account for the various factors that can affect discard survival (e.g., Benoît *et al.* 2010; 2012; Davis, 2010).

The third aim of the study was to better understand how discard impacts might be mitigated. While increased selectivity can reduce the catch of non-marketable organisms, reducing fishing impact on them (Broadhurst *et al.*, 2006), certain modifications to fishing operations also have the potential to increase discard survival rates of the organisms for which catch is unavoidable in a particular fishery (e.g. Benoît *et al.*, 2010; Enever *et al.*, 2010). Consequently, understanding the technical, environmental and biological factors that affect discard survival is key to developing effective discard mortality mitigation measures. Main and interacting effects on survival are summarized in Davis

(2002). Technical factors relate to capture stressors from different gear types and deployments (e.g. haul duration, catch handling, and so on.). Environmental conditions can induce additional stress through changes in e.g. salinity, air and sea surface temperature (Harris and Ulmestrand, 2004; Uhlmann and Broadhurst, 2013b). Among the biological factors, especially the size and physiology to withstand stress and injury are important.

5.3 Materials and methods

5.3.1 *Discard survival experiments*

Survival experiments were performed aboard the RV 'Belgica' during a total of six five-day fishing trips in the southern North Sea (ICES subarea IVc, ICES statistical rectangles 31F1, 31F2, 32F1, 33F1, 33F2 and 34F1) (Table 5.1)¹⁰. This area was selected because of its importance for the Belgian beam trawler fleet (Chapter 4). Fishing was conducted on commercial fishing grounds, based on tracks provided by commercial fishermen. Two 4 m beam trawls were attached next to each other with an extra trawl head in the middle of the 8 m beam to allow catch comparison trials as part of another study (e.g. Fonteyne and Polet, 2002). Organisms were retrieved from one of the beam trawls equipped with a chain mat and an 80 mm diamond mesh codend. The net was made of polyethylene netting, single braided in the top panel and double braided in the lower panel. The cod-end was made of double braided polyethylene netting with a twine thickness of 4 mm. The duration of experimental treatment hauls ranged between 90 and 100 min (interquartile range) with a speed of about 4 knots and the length of the fishing warps was about 2.5 to 3 times the depth, consistent with practices in the commercial "eurocutter" fishery. We performed 35 hauls to collect fish, and 25 to sample benthic invertebrates. Five hauls of limited duration (< 20 min) and with minimal post-haul handling times of captured fish before being placed in holding tanks (<5 min) were conducted to serve as experimental controls. A total of 26 and 48 individuals of sole and plaice were caught in these control hauls.

The organisms from experimental treatment hauls were deposited on deck, collected in boxes to be weighed, after which specimens were placed in holding tanks. Previous studies have shown that air exposure is one of the greatest contributors to discard mortality within and among species (Benoît *et al.* 2013 and references herein). In the commercial fisheries of the North Sea and the Western English Channel, the mean duration of air exposure (+/- S.E.), measured as the difference between the release of the catch on deck and the first and last organism being returned to the sea, varies

¹⁰ The mean catch composition was summarized in Figure A5.1.

between a minimum of 12 (+/-5) min and a maximum of 30 (+/-8) min. Air exposure ranged between 15 and 20 min in the experiments (interquartile range). Fatally damaged specimens were registered but not held, while others were held in two types of plastic holding tanks. A maximum of 8 roundfish or skate individuals were kept in holding tanks of 175 L (70*50*50 cm). Flatfish and invertebrates were held in 24 L tanks (60*40*11 cm) as used in Van Beek *et al.* (1990). A maximum of 4 flatfish, 20 invertebrates or 6 edible crabs were retained per tank. A continuous flow of fresh, ambient sea surface water was provided. No feeding took place during the subsequent observation period.

Survival was monitored twice daily for a minimum duration of 60 h in the holding tanks, an artificial limit imposed by the trip durations. Fish mortality was based on common death signs, such as motionlessness during approximately 2 min of observation, non-response to physical stimuli, flaring of the gills (roundfish), curling of wings (skates), and rigor mortis. Invertebrates were considered dead in the absence of body movements, e.g. a live sea mouse curls its body by contracting longitudinal muscles upon touching (Kaiser and Spencer, 1995). Dead specimens were removed from the holding tanks.

Explanatory factors for survival were recorded (Table 5.2). Haul-specific environmental characteristics (mean and range) were catch weight (137.6 kg, 40-414.5 kg), mean depth fished (30.3 m, 10-50 m), salinity (34.5 PSU, 33.3-35.0 PSU), SST (9.6 °C, 4.6-12.4 °C) and air temperature (7.9 °C, 3.6 °C-11.4 °C). Length and physical damage were considered important at the level of the individual. Invertebrates were not measured. Fish length was measured to the nearest cm. Physical injuries were assessed according to taxon-specific schemes. Benthic invertebrates were assessed following the scheme of Veale *et al.* (2001) (Table 5.2). Sea mouse (*Aphrodita aculatea*) was categorised as either undamaged or damaged (crushed). The physical injuries of fish species were evaluated by the Catch Damage Index (CDI), in order to avoid subjectivity of categorising fish by inspecting their condition and behaviour (e.g. Van Beek *et al.*, 1990). CDI was originally developed to assess quality defects caused by fishing gears (Esaïassen *et al.*, 2013), but was modified to evaluate relevant physical damage for fish survival (Table 5.3). The intention was to reduce subjectivity as much as possible by accounting for purely measurable physical damages as a predictor of survival.

Table 5.1 Summary of environmental, technical and biological data collected during six five-day fishing trips by RV 'Belgica' in the southern North Sea.

Fishing trip	24-28 November 2008	1-5 December 2008	2-6 February 2009	30 March – 3 April 2009	26 – 30 November 2009	30 November – 4 December 2009
<i>ICES Rectangles fished</i>	31F2, 33F1 and 34F1	32F1 and 33F1	31F1, 32F1, 33F1, 33F2	31F2	33F1	33F1
<i>Depth fished (m)</i>	14 – 35	25 - 33	28 – 50	10 - 13	15 - 32	28 - 33
<i>Salinity (PSU)</i>	33.8 (0.4)	34.0 (0.1)	35.0 (0.0)	35.0 (0.0)	34.9 (0.1)	34.2 (0.2)
<i>Sea surface temperature (°C)</i>	9.3 (0.8)	9.3 (0.4)	5.8 (0.4)	8.2 (0.1)	12.1 (0.1)	11.3 (0.1)
<i>Air temperature (°C)</i>	7.5 (0.9)	5.2 (1.0)	4.9 (0.5)	9.3 (0.6)	10.84 (0.37)	9.43 (0.54)
Commercial hauls						
<i>Catch weight (kg)</i>	40.0 - 414.5	67.2 - 197.0	70.9 - 200.6	64.5 - 251.9	70.9 - 200.6	64.5 - 153.0
<i>Haul duration (min)</i>	97.0 (9.7)	99.3 (8.9)	91.7 (4.1)	87.0 (6.1)	84.2 (16.9)	91.3 (3.5)
<i>Number of hauls</i>	5	7	6	9	7	5
<i>Number of individuals:</i>						
<i>Gadus morhua</i>	17	9	-	27	3	8
<i>Merlangius merlangus</i>	-	30	-	-	26	20
<i>Pleuronectes platessa</i>	-	5	24	47	9	12
<i>Rajidae</i>	-	34	38	-	39	44
<i>Solea solea</i>	42	48	52	24	48	56
<i>Trisopterus sp.</i>	-	31	-	-	13	5
<i>Total length (cm)</i>						
<i>Gadus morhua</i>	32 - 52	33 - 63	-	32 - 56	54 - 63	42 - 75
<i>Merlangius merlangus</i>	-	17 - 35	-	-	17 - 36	15 - 28
<i>Pleuronectes platessa</i>	-	20 - 27	17 - 29	15 - 31	17 - 29	18 - 32
<i>Rajidae</i>	-	10 - 43	14 - 46	-	8 - 53	7 - 47
<i>Solea solea</i>	13 - 35	16 - 31	17 - 31	19 - 33	15 - 28	15 - 28
<i>Trisopterus sp.</i>	-	13 - 30	-	-	15 - 28	16 - 29
"Reference" hauls						
<i>Number of hauls</i>	-	-	1	2	-	2
<i>Number of individuals:</i>						
<i>Pleuronectes platessa</i>	-	-	1	24	-	-
<i>Solea solea</i>	-	-	8	18	-	20
<i>Total length (cm)</i>						
<i>Pleuronectes platessa</i>	-	-	16	18-31	-	-
<i>Solea solea</i>	-	-	21-34	23-34	-	19-29

Table 5.1 (continued). Summary of environmental, technical and biological data collected during six five-day fishing trips by RV 'Belgica' in the southern North Sea.

Fishing trip	24-28 November 2008	1-5 December 2008	2-6 February 2009	30 March – 3 April 2009	26 – 30 November 2009	30 November – 4 December 2009
Commercial hauls						
<i>Asterias rubens</i>	103	40	75	-	-	-
<i>Ophiura sp.</i>	-	-	27	20	1	7
<i>Psammechinus miliaris</i>	59	60	-	-	-	-
<i>Cancer pagurus</i>	5	5	8	-	-	22
<i>Liocarcinus sp.</i>	18	-	29	21	-	5
<i>Pagurus bernhardus</i>	13	42	17	-	-	-
<i>Aphrodite aculeata</i>	-	-	25	-	15	-

Table 5.2 Scores for damage inflicted on benthic invertebrates (modified from Veale *et al.*, 2001).

Species	1	2	3	4
<i>Starfish / Brittlestars</i>	No visible damage	Arms missing	Worn and arms missing / minor disc damage	Major disc damage / crushed
<i>Crabs</i>	No visible damage	Legs missing / small carapace cracks	Major carapace cracks	Crushed
<i>Hermit crabs</i>	No visible damage	Edge of shell chipped	Shell cracked or punctured	Crushed
<i>Urchins</i>	No visible damage	<50 % spine loss	>50 % spine loss / minor cracks	Crushed

Table 5.3 Modified Catch Damage Index (CDI) to evaluate physical injuries for fish after catching and handling operations (modified from Esaiassen *et al.*, 2013). Bruises are scored separately for head, body and tail.

Catch Damage Index	Description	Score
<i>Gear related damages</i>	No gear marks	0
	Gear marks such as incisions	1
<i>Skin-abrasion</i>	<10 % scale loss	0
	Between >=10 % and <50 % scale loss	1
	>=50 % scale loss	2
<i>Bruises (separate scoring for head, body and tail)</i>	Non discoloration	0
	<50 % discoloration on the area	1
	>=50 % discoloration on the area	2
<i>Pressure injuries</i>	No compression detected	0
	<30 % compression detected	1
	>=30 % compression detected	2
<i>Broken spine</i>	No	0
	Yes	1
<i>Fin and tail damage</i>	No marks	0
	<30 % visible marks	1
	>=30 % visible marks	2
<i>Max total score (CDI)</i>		14

5.3.2 Discard survival analysis

5.3.2.1 Estimating empirical survival functions

The survival of a taxon was estimated as a function of time using the non-parametric Kaplan-Meier procedure (Kaplan and Meier, 1958). The Kaplan-Meier survival probability for each finite time interval i is defined as:

$$\hat{S}(t) = \prod_{t_i \leq t} \left(\frac{n_i - d_i}{n_i} \right) \quad [1]$$

where $\hat{S}(t)$ is the estimated survival probability at time t , n_i is the number of individuals at risks of death during the finite time interval i and d_i is the number of deaths during i . This approach easily accommodates right-censored observations, which are organisms for which death was not observed because the study was terminated before they died.

5.3.2.2 Predicting survival based on physical injuries

The ability of the CDI to predict fish survival was assessed intraspecifically using the modelling approach developed by Benoît *et al.* (2012), in which a small number of parametric survival models with CDI (or analogous covariates) as a predictor were fitted to the experimental data. The classes of the CDI were recoded into a binary “injury” variable, with all values below the overall mean CDI in one class and those above in the other. This resulted in sufficient observations to fit all models for both injury classes, and increased the discriminating association with survival, (e.g. Enever *et al.*,

2008). Estimation of survival during captivity was based on the analysis of survival probability as a function of time, while accounting for right-censored data. The underlying distribution was a Weibull-type survival function, conditional on the injury class:

$$\hat{S}(t) = \exp[-(\alpha * t)^\gamma] \quad [2]$$

where $\hat{S}(t)$ is the estimated survival probability and the scale and shape parameters of the Weibull distribution are given by α and γ . The survival model [2] assumes that all individuals follow the same survival function, and that $\hat{S}(t)$ is a continuous declining function of t . The homogeneity assumption is violated if only a portion of the captured fish is adversely affected by the capture and handling process, such as the most severely injured individuals. Benoît *et al.* (2012) accounted for inhomogeneity using a two component survival mixture model (SMM). The first component models the survival of individuals adversely affected by trawling (eq. [2]), which are assumed to follow a common survival function while the second component accounts for the proportion of individuals that were not adversely affected, which are assumed to not be at risk of dying during the course of the captivity study (for more details and justification, see Benoît *et al.*, 2012). The resulting survival mixture model (SMM) is:

$$\hat{S}'(t) = \pi * \exp[-(\alpha * t)^\gamma] + (1 - \pi) \quad [3]$$

where π is the proportion of individuals that were adversely affected by the fishing event. When all individuals are adversely affected, i.e. $\pi = 1$, the equation equals eq. [2]. Covariates suspected of affecting the survival probability can be included in the definition of α (i.e. covariates affect the rate of mortality over time) and/or π (i.e. covariates affect the probability that an individual is affected or not). Six models were defined by varying how covariates were incorporated into the parameters α and π (Table 5.4). The relative evidence for each of these models was assessed using differences in Akaike's Information Criterion corrected for small sample sizes, ΔAICc . Models with $\Delta\text{AICc} < 2$ were interpreted as having similar support in the data, while ΔAICc values between 3 and 7 suggested less support for the competing model, and values > 10 suggested the alternative model being unlikely (Burnham and Anderson, 2002). The fit of the SMM selected via ΔAICc was assessed by comparing model predictions to the empirical Kaplan-Meier (KM) survival curves, which do not assume an underlying survival function. The SMM model selection was considered suitable if the selected models fitted well within the 95 % confidence intervals of the KM curves. The difference in KM curves between injury classes was also tested by the formal rank test for right-censored survival data (Harrington and Fleming, 1982).

Table 5.4 Assumption for the parameters α and π in equation [3], to define the six competing models for the analysis of the fish survival probability. X is a matrix of the injury classes, with β , β_1 and β_2 being the vectors of parameters for each injury class. The entry 'constant' indicated that α and/or π were estimated by model fitting (developed by Benoît *et al.*, 2012).

Model	α	π	Interpretation
Weibull 1 (W1)	Constant	1	Common survival function for all fish
Weibull 2 (W2)	$\exp(X'\beta)$	1	Common survival function for each injury class (Eq. 2)
Mixture 1 (M1)	Constant	Constant	Common survival function for a fixed proportion of affected individuals
Mixture 2 (M2)	$\exp(X'\beta)$	Constant	Common survival function within each injury class for a fixed proportion of affected individuals
Mixture 3 (M3)	Constant	$[1 + \exp(X'\beta)]^{-1}$	Common survival function for affected individuals, with the proportion affected dependent on injury class
Mixture 4 (M4)	$\exp(X'\beta_1)$	$[1 + \exp(X'\beta_2)]^{-1}$	Common survival function within each vitality class, where the proportion of affected individuals also depends on injury class

Effective discard mortality studies should possess two key features (e.g., Davis 2002; Broadhurst *et al.* 2006). The first is the use of proper control subjects to account for mortality induced by captivity, as was included in this study (see above). The second is to ensure that the duration of captivity is sufficiently long to cover potential delayed mortality of individuals resulting from the capture and handling. An advantage of the SMM approach is that the potential for delayed mortality can be assessed. Evidence for models W1 and W2 (Table 5.4) implies on-going mortality of individuals, while evidence for the remaining models (mixture models M1-M4) implies that survivorship in the samples will level off at $\hat{S}^i(t) \approx 1 - \pi$. Furthermore, if there is evidence for the mixture models, the time at which survivorship is expected to level off (say when 99 % of affected individuals have died), t_s , can be estimated as:

$$t_s = \frac{-\ln(0.001)^{(1/\gamma)}}{\alpha} \quad [4]$$

As such, we calculated t_s for the species in our study to determine what the ideal duration of captivity would have been, assuming the model is correct. These values can then serve in planning future survival studies involving the captivity of these species.

In addition to the intraspecific survival models, CDI scores were compared interspecifically with a non-parametric Kruskal-Wallis test and posthoc comparison by Mann-Whitney tests with false discovery rate (FDR) corrections (Benjamini and Hochberg, 1995; García *et al.*, 2004).

Evidence for possible relationships between survival and the damage index for invertebrates was tested using the Somers' D statistic. This is a non-parametric asymmetric, rank biserial correlation coefficient, which can handle ties (Somers, 1962). The causal association between the ordinal damage scores and death (treated as a binary variable) was estimated and was reported as the

confidence intervals (CI) around the Harrell's C-index, which parameterizes Somers' D to a probability scale from 0 to 1 (Harrell *et al.*, 1982). In contrast to the case for fish, this approach was chosen because an absence of individuals in particular damage categories and particular properties of the data, such as most or all mortality occurring prior to holding for certain species, would have required a more complex application of the “fish” models. Instead a more simple analysis was used to determine the role of injuries on survivorship and an examination of survivorship at contrasting short and long holding durations was used to summarize the evidence for delayed mortality.

5.3.2.3 Factors potentially affecting fish discard survival

The potential contribution of catch weight, fish length, CDI, fishing depth, salinity, SST, air temperature and fishing trip to survival variability was investigated if more than 20 hauls were available, which restricted the analysis to sole (N=32) and skates (N=22). The discretized physical injury classes were re-examined, both as a single factor and in interaction with weight or length, which were considered to be the potentially strongest of possible interactions. Collinear explanatory variables were removed from the analysis according to a variance inflating factor (VIF) of two, to enable the detection of weak ecological signals (Zuur *et al.*, 2010). We used a mixed-effects Cox proportional hazards model (Therneau and Grambsch, 2000) of the form:

$$\hat{h}(t) = h_0(t) \exp(X'\beta + Z'b) \quad [5]$$

where hazard function $\hat{h}(t)$ is the probability of mortality at time t , conditional on survival until time t . In the model, $\hat{h}(t)$ is conditional on a set of predictor covariates X' (catch weight, depth, length, injury class, fishing trip and SST in the case of sole) and a Gaussian haul-specific random effect Z' . The Cox model is a semi-parametric method in which hazards are estimated from the ranks of mortality times (Cox and Oakes, 1984). As defined above, the model deals with proportional hazards, in that the hazard for an individual at time t is a fixed proportion of the hazard of any other individual that depends on their state with respect to the covariates and the covariate parameter values. Parameters were estimated by partial maximum likelihood (Ripatti and Palmgren, 2000). In contrast to the SMMs, the Cox model does not assume any particular baseline function h_0 . While the absence of a parametric hazard function prevents a direct mechanistic interpretation of survival patterns such as provided by the SMMs, it provides a very flexible model in which to test for the effects of covariates without having to worry about the appropriateness of a particular parametric form. Cox regression was exclusively used for determining the potential influence of factors on survival. Model selection was based on the $\Delta AICc$.

5.4 Results

5.4.1 Discard survival

The survival probability during the first part of the observation period was high for all invertebrate species (>90 %), except for swimming crabs (Figure 5.1, Table 5.5). The survival of swimming crabs was 78 % during the first 24 h. Approximately half of all individuals were monitored for at least 60 h. The Kaplan-Meier survival estimates for these longer observation periods did not decrease considerably for most of the species, except for green sea urchins. The survival of the latter was 75 % after 72 h observation period.

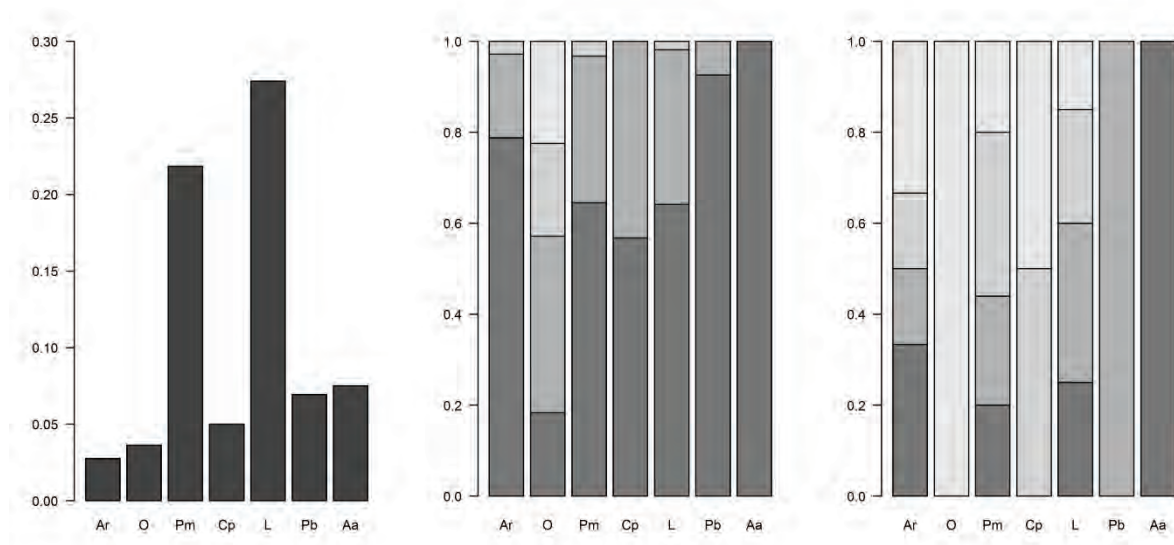


Figure 5.1 The percentage of dead organisms at the end of the observation period (left), and the relationship between the physical injuries for surviving (middle) and dead (right) organisms. The physical injury classes (Table 5.1) are distinguished using grey shading, from dark grey for less severe physical injuries (class 1) to light grey for the most severe injuries (class 4). Aa: *Aphrodita aculeata*, Ar: *Asterias rubens*, Cp: *Cancer pagurus*, L: *Liocarcinus* sp., O: *Ophiura* sp., Pb: *Pagurus bernhardus*, and Pm: *Psammechinus miliaris*.

The survival probability of the six examined fish taxa was much lower (Table 5.6). All whiting (*Merlangius merlangus*) and pouting (*Trisopterus* sp.) died within 24 h. Only 14 % of sole (*Solea solea*) survived to 91 h of observation, while 48 % of plaice (*Pleuronectes platessa*) survived to 77 h. In contrast, the survival probability was higher for cod (*Gadus morhua*) (66 % to 88 h) and skates (*Rajidae*) (72 % to 80 h). The Kaplan-Meier survival estimates of short hauls (<20 min) was 100 % for plaice (25 individuals) and 96 % for sole (46 individuals).

Table 5.5 Kaplan-Meier survival estimates with standard errors (S.E.) for benthic invertebrates during a short and long observation period. The number of investigated individual, N, and number of dead organisms, N(dead), are indicated for each time interval. Irregular times were due to the variable times at which individuals could be collected from consecutive hauls and the subsequent irregular steps at which occurrence of mortality events could be registered.

Species	N	Time (h)	N(dead)	Survival probability (%)
<i>Asterias rubens</i>	218	17	3	99 (1)
	107	65	6	97 (1)
<i>Ophiura sp.</i>	51	22	2	96 (3)
	24	60	2	96 (3)
<i>Psammechinus miliaris</i>	119	38	3	98 (2)
	83	72	23	80 (5)
<i>Cancer pagurus</i>	39	32	2	95 (4)
	20	84	2	95 (4)
<i>Liocarcinus sp.</i>	73	24	16	78 (6)
	53	67	18	75 (7)
<i>Pagurus bernhardus</i>	72	53	3	96 (3)
	43	72	4	94 (3)
<i>Aphrodita aculeata</i>	40	34	2	92 (5)
	25	69	3	92 (5)

Table 5.6 Kaplan-Meier survival estimates with standard errors (S.E.) for six fish species held in holding tanks after commercial hauls (1.5 h) with a chain mat beam trawl. Survival of plaice and sole was also tested for short hauls, which served as a control subjects for the experiments. Censored individuals are those that survived the entire holding period. The number of investigated individuals, N, and number of dead organisms, N(dead), are indicated for each time interval. Irregular times were due to the variable times at which individuals could be collected from consecutive hauls and the subsequent irregular steps at which occurrence of mortality events could be registered.

	N	Time (h)	N(dead)	Survival probability (%)
Commercial hauls				
<i>Rajidae</i>	141	65	34	77 (5)
	108	80	40	72 (6)
<i>Gadus morhua</i>	64	34	18	72 (8)
	45	88	21	66 (9)
<i>Merlangius merlangus</i>	76	21	76	0 (-)
<i>Trisopterus sp.</i>	49	16	49	0 (-)
<i>Pleuronectes platessa</i>	97	57	30	69 (7)
	88	77	41	48 (15)
<i>Solea solea</i>	246	64	186	29 (10)
	208	91	202	14 (25)
“Reference” hauls				
<i>Pleuronectes platessa</i>	25	63	0	100 (-)
	12	87	0	100 (-)
<i>Solea solea</i>	46	52	2	96 (3)
	29	63	2	96 (3)

5.4.2 *Survival with respect to physical injuries*

There was a strong and statistically significant ($p < 0.0001$) association between the number of dead organisms and the physical injuries for hermit crabs ($C = 0.96$; $CI = [0.95, 0.97]$), ophiurids ($C = 0.89$; $CI = [0.87, 0.90]$) and green sea urchins ($C = 0.81$; $CI = [0.79, 0.83]$) (Figure 5.1). The association was less strong for common starfish ($C = 0.78$; $CI = [0.75, 0.81]$; $P < 0.05$) and swimming crabs ($C = 0.76$; $CI = [0.73, 0.79]$). Furthermore, the only two dead edible crabs were respectively classified within damage classes 3 and 4. There were no physical injuries detected for sea mouse.

The mean CDI of all fish taxa (+/- SD) was 2.94 (+/- 1.78), being highest for cod (3.74) and lowest for skates (2.57). The variability of the CDI was highest for plaice (SD = 1.99), cod (1.98) and skates (1.96), while physical injuries varied less for sole (1.67), whiting (1.20) and pouting (1.15) (Table 5.7). A Kruskal Wallis test revealed significant interspecific differences ($\chi(5) = 19.7$, $P < 0.01$). Posthoc tests showed that CDI scores of cod differed significantly from those of skates ($P < 0.01$, $r = 0.26$), whiting ($P < 0.01$, $r = 27.3$), sole ($P < 0.05$, $r = 15.2$) and plaice ($P < 0.1$, $r = 16.8$). CDI scores of skates were different from sole ($P < 0.1$, $r = 11.1$) and pouting ($P < 0.1$, $r = 15.7$).

The model selection for skates and plaice highlighted a difference in survival between different injury classes as differences in AICc were high (>10) for models in which survival probability was unrelated to injury (models W1 and M1) (Table 5.8). The “best” model for plaice and skate was a simple Weibull model with survival rate as a function of physical injuries (model W2). However, $\Delta AICc$ were small for models M3 and M4, indicating some evidence for mixture models in which the mixture weight π was a function of injury. KM curves for both species matched well with the predicted survival functions, e.g. W2 and M3 models for skates and W2 model for plaice (Figure 5.2). A formal rank test for right-censored survival data confirmed a significant difference in survival probability between injury classes for both plaice ($\chi^2_{(1, N=96)} = 20.7$, $p < 0.0001$) and skates ($\chi^2_{(1, N=147)} = 26.9$, $p < 0.0001$).

The model selection procedure for cod was less clear cut, with support for several of the competing models. Although the mean CDI of cod was highest amongst taxa, and had a high degree of variation, there was evidence for models with (W2 and M3) and without (W1) an effect of injury on survival. The KM curves of both injury classes showed a high degree of overlap (Figure 5.2), and a statistical difference was not detected ($\chi^2_{(1, N=61)} = 1.2$, $p = 28$).

For sole, M4 was selected as the most likely model, although there was also evidence for model M3 (Table 5.8; Figure 5.2). Though this provided some support for the relationship between physical injuries and survival of sole, $\Delta AICc$ values indicated that other models excluding this relationship

were not improbable. The SMM curves and the KM confidence intervals visualized the poor discriminative power of physical injuries to predict differences in survival probability (Figure 5.2). Indeed, no statistical difference was detected between the survival of different injury classes ($\chi^2_{(1, N=260)} = 0.1, p = 0.82$).

The time at which survivorship is expected to level off, t_s , was estimated from the M3 models. The predicted ideal duration of captivity required to observe all experimental mortality was approximately 5 days for sole ($t_s = 4.7$; CI = [3.6, 5.9]), 9 days for skates ($t_s = 8.7$; CI = [5.1, 12.3]) and plaice ($t_s = 8.9$; CI = [4.8, 13.1]) and 21 days for cod ($t_s = 21.1$; CI = [0, 61.2]).

Table 5.7 Frequency of the CDI scores across the observed fish species.

Species	Catch damage index										
	0	1	2	3	4	5	6	7	8	9	10
<i>Rajidae</i>	22	28	25	32	17	13	5	1	3	-	1
<i>Gadus morhua</i>	-	5	14	15	10	6	5	4	1	2	-
<i>Merlangius merlangus</i>	2	15	14	26	14	3	-	-	-	-	-
<i>Trisopterus</i> sp.	-	3	16	13	11	6	-	-	-	-	-
<i>Pleuronectes platessa</i>	9	17	18	14	17	10	8	3	-	1	-
<i>Solea solea</i>	15	41	68	48	42	40	8	5	1	-	-

Table 5.8 Selection of survival mixture models based on $\Delta AICc$ values. See Table 5.4 for the definition of each of the six competing models. The “best” models have zero $\Delta AICc$ values, indicated in bold.

Species	Candidate models					
	W1	W2	M1	M2	M3	M4
<i>Rajidae</i>	19.9	0	58.4	60.4	1.3	3.4
<i>Gadus morhua</i>	0	1.5	3.4	3.8	1.9	5.3
<i>Pleuronectes platessa</i>	18	0	16.7	24	3	4.1
<i>Solea solea</i>	6.1	7.4	8	8.7	2.4	0

Table 5.9 Factors affecting fish survival: parameter estimates with standard errors (S.E.) and p-values for the final mixed effects Cox models for sole and skates.

	Exp(estimate)	S.E.	p-value
<i>Rajidae</i>			
Injury class	9.049	0.373	<0.0001
Length	0.891	0.023	<0.0001
<i>Solea solea</i>			
Catch weight	1.006	0.003	0.020
Depth	1.028	0.014	0.048
Injury class	1.467	0.179	0.033
Length	0.936	0.024	0.005

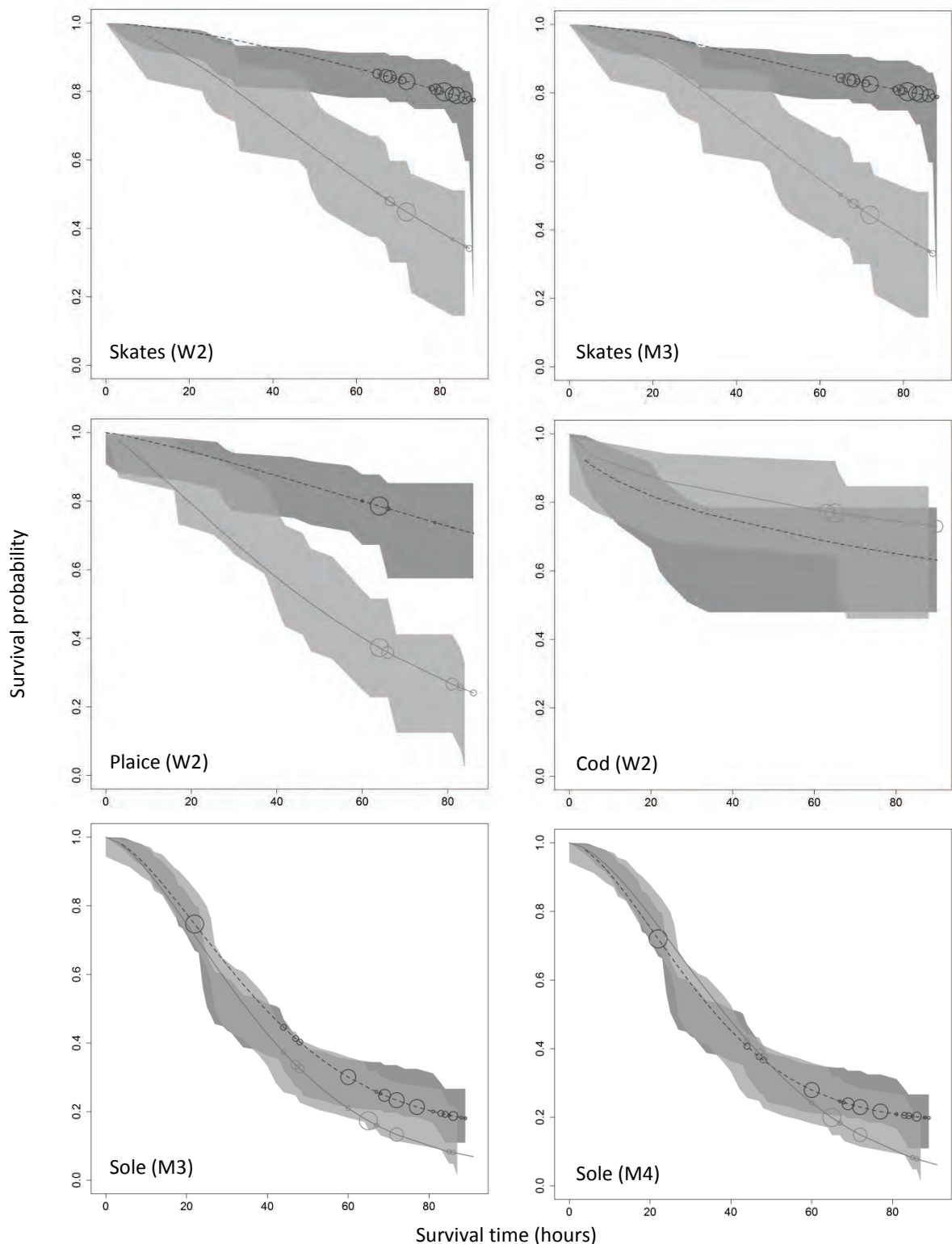


Figure 5.2 Survival probability of skates (upper left: W2, upper right: M3), plaice (middle left: W2), cod (middle right: W2), and sole (lower left: M3, lower right: M4) in relation to time (h) in the holding tanks. Individuals with limited physical injuries are indicated in dark grey, except for cod. Lines are the fits from the Survival Mixed Models, while shaded areas are point-wise 95 % confidence intervals from the Kaplan-Meier survival analysis. The magnitude of right-censored individuals is indicated by the size of the circles along the fits.

5.4.3 *Factors potentially affecting discard survival*

The contribution of catch weight, fish length, physical injuries, salinity, SST, air temperature and fishing trip was examined for the short-term survival probability of sole and skates. VIFs were <2 for depth, SST, catch weight, length, CDI and fishing trip, which were selected for the modelling procedure for sole. Salinity and air temperature correlated significantly ($P < 0.0001$) with fishing trip (Respective Pearson $r^2 = 0.89$; 0.55). The “best” model for sole (lowest AICc) retained catch weight, depth, injury class, length, fishing trip and the interaction between injuries and weight. This model explained 20.7 % of the overall variability, including two variables that were insignificant ($P = 0.051$ for depth; $P = 0.052$ for the interaction). The next “best” model was more parsimonious, explained 17.8 % and retained only significant variables: catch weight, depth, length and physical injuries. Table 5.9 summarises the results for the latter model. A higher catch weight, a greater depth and more injuries decreased the survival probability significantly ($P < 0.05$). Larger individuals had significantly better survival probabilities ($P = 0.005$). In contrast, 69.0 % of the variability in survival probability of skates was explained by a model that included length and physical injuries. Depth, catch weight, length, CDI and fishing trip were selected for the candidate models ($VIF < 2$). Salinity correlated significantly with fishing trip (Pearson $r^2 = 0.97$, $P < 0.0001$), depth was correlated to SST (Pearson $r^2 = -0.73$, $P < 0.0001$) and air temperature (Pearson $r^2 = -0.48$, $P < 0.0001$). Length and injuries were selected with a strong preference in eight candidate models ($\Delta AICc > 20$ for the other models). Those models contained catch weight, fishing trip or the interaction of length and injury class, a combination of those or none of them. As was the case for sole, the inclusion of additional variables did not always coincide with adding a significant factor or increasing explanatory power of model variability considerably (<6 %). Therefore the most parsimonious model was reported, i.e. with length and injuries (Table 5.9). Survival probability was significantly higher for larger individuals and less injured skates ($P < 0.0001$). The length and injury effect was higher for skates than for sole (Table 5.9). The hourly hazard of death decreased with length by a factor 0.936 for soles and 0.891 for skates. In other words, the hourly survival chances of skates during the observation period increased by 10.9 % with length, while with 6.4 % for sole. Hourly survival probabilities decreased with increasing CDI scores by almost a tenfold more for skates than for soles.

5.5 Discussion

5.5.1 Discard survival estimates from tank-based experiments

The survival of discarded invertebrates in the 4 m beam trawl fishery with chain mats (Table 5.5, Figure 5.1, this study) followed the generally observed pattern in previous studies (Table A5.1). All short-term survival estimates for Asteroidea and Gastropoda were consistently very high (>85 %), which is in agreement with other bottom trawling studies (e.g. Bergmann and Moore, 2001). A great deal of variation was observed in discard survival between species of Crustacea (14-93 %) and Bivalvia (10-98 %) (e.g. *Liocarcinus* sp. and *Cancer pagurus*, this study). Species-specificity is expected to be due to differences in biological traits (fragility, flexibility, physiology, and so on). Hard-bodied species without limbs (e.g. whelks) seem to have a better short-term survival rate than species with fragile externalities (e.g. swimming crabs) (Kaiser and Spencer, 1995). This is supposedly the reason why physical injuries show such a good association with invertebrate discard survival (this study) and why catch composition and weight help when predicting survival (Bergmann *et al.*, 2001a). In addition, survival of invertebrate discards also depends on species-specific differences in secondary responses, e.g. tissue damage results in marked physiological stress responses. As an example, *Liocarcinus* sp. and *Cancer pagurus* both induce a rapid rise in glucose and lactate to autotomize limbs (Bergmann *et al.*, 2001b; Patterson *et al.*, 2007). Differences in secondary responses between both species may contribute to the observed differences in survival probability of this study, by analogy with marked differences in autotomy reflex after trawling for *Liocarcinus depurator* and galatheids (Bergmann and Moore, 2001). Further, high within species variability has been reported, e.g. survival rate of brown shrimp (*Crangon crangon*) can vary between 30 % and 80 % in different environments (Gamito and Cabral, 2003; Lancaster and Frid, 2002). Other plausible contributors to the observed differences are thus technical and environmental conditions, such as temperature shock (Raicevich *et al.*, 2011), salinity (Harris and Ulmestrand, 2004), etc. While physical injuries of benthic invertebrates were clearly associated with survival probabilities in this study, causal relationships could not be established and would require a more complex application of the “fish” models, which could not be performed due to limitations in the collected invertebrate data (Table 5.1).

Discard survival for fish was generally lower than that of invertebrates, though there are generally differences amongst fish taxa. For example, in a cross-species study, Benoît *et al.* (2013) found that Rajiformes generally have a higher survival potential than Pleuronectiformes, followed by Gadiformes. While Benoît *et al.* (2013) used a proxy for survival (time-to-mortality), their findings on elasmobranchs are confirmed by estimates from tank-based experiments for various bottom otter

trawl fisheries (Enever *et al.*, 2008; Laptikhovsky, 2004; Benoît *et al.*, 2012; Mandelman *et al.*, 2013). This study provides the first estimates for beam trawl fisheries, confirming a relatively high skate survival rate (>70 %), which is relatively close to the estimate by Enever *et al.* (2008) for the UK otter trawl fishery (59.1 %). The Pleuronectiformes in this study, sole and plaice, had a survival rate of 48.2 % and 13.9 % at 77h and 91 h post capture respectively. These values compare favourably with those obtained from studies involving tickler chain beam trawls for sole and with chain mat beam trawling for plaice. Our estimates are considerably higher than those of plaice survival in tickler chain beam trawling and lower for sole in chain mat beam trawling in the western English Channel (Table A5.1). Discrepancies in results between studies may stem in part from a lack of fully quantifying post-capture mortality. Unless studies are terminated only once mortality of held fish has stabilized, discard mortality is likely to be underestimated to varying degrees. This may be the case for plaice, for which mortality had yet to stabilize when both our study and that of van Beek *et al.* (1990) were terminated. Due to their low number, individuals of dab (*Limanda limanda*) and lemon sole (*Microstomus kitt*) were not analysed in detail here, but also indicate that these Pleuronectiformes suffer high discard mortalities. From the 15 individuals of dab, 6 were dead after 60 h observation with 7 individuals censored. Only 3 out of 25 lemon soles were alive after a 60 h observation period. Surprisingly, the expected low survival rates for Gadiformes (Lindeboom and de Groot, 1998) were not fully confirmed. Indeed, whiting and pouting did not survive a short observation period, but the survival probability for cod was considerably higher than expected (65.9 % at 88h). The limited fishing depth for catching the cod individuals may be a plausible explanation. All individuals were caught at depths between 10 and 33 m, which is expected to result in higher cod survival due to less barotrauma during capture compared to capture at a greater depth (Pálsson *et al.*, 2003; Van der Kooij *et al.*, 2007).

The discard estimates clearly demonstrate different survival probabilities between taxa, but the absolute survival probabilities should be interpreted with caution. Benoît *et al.* (2012) explained that a multitude of conditions determine post-release survival (e.g. fishing depth, temperatures, handling practices, etc.) and that deriving estimates that are relevant to a fishery requires integration of the relevant conditions experienced by fish captured and discarded in that fishery. While the conditions experienced by the fish in our study were consistent with the types of condition present in the fishery, it is unlikely that they properly represented the distribution of conditions experienced by discarded fish in the fishery. Furthermore, the predictions of survival probability are based on a short term study in non-natural conditions and are likely overestimated since mortality due to post-release infection and predation risk was not quantified. Increased predation risk can be due to impaired swimming abilities (e.g., due to distended swim bladders) or due to post-traumatic behaviour, which

can last for weeks beyond the period of observation. This was for instance observed *in situ* (North Sea) from data-storage tags in trawl captured cod, for which natural pre-capture vertical movements were only re-established after 10 days of being returned to the sea (Neat *et al.*, 2009). Survival estimates from tank experiments should therefore be used as a step into a broader framework of understanding mortalities from discarding.

This study and others have highlighted the need to fully quantify capture and handling mortality by ensuring there is no delayed mortality in the study (i.e., that mortality has levelled off before the study is terminated). The SMM used here and in Benoît *et al.* (2012) have the advantage of essentially testing whether mortality has stabilized and allowing investigators to estimate the time at which this occurs. Some authors have suggested that survival curves generally level off within 4 days (Wassenberg and Hill, 1993), which is also suggested as the minimal period of inhibition of a species' normal activity (Neat *et al.*, 2009). These findings were confirmed by the time of levelling off for sole at approximately 5 days. However, the survival only stabilized over a time of approximately 9 days for skates and plaice, and 21 days for cod. Evidence for on-going mortality beyond a study's duration was also found in the study of Benoît *et al.* (2012) for witch flounder and for 9 out of 29 cases in the authors' review of several long-term discard and escape mortality studies. The long observation period for cod was however in contrast to the results of Benoît *et al.* (2012), where an ideal duration of captivity of 2 days was suggested. The reason for the long observation period in this study is due to 3 cod individuals that died late in the period of captivity (> 60 h). When those individuals were artificially excluded from analysis, an ideal duration of captivity was 1.9 days ($t_s = 1.9$; [CI= 1.2, 2.8]). Our study hence confirms the rapid mortality for cod, but also provided support for the potential of a delayed mortality.

The factors discussed above are likely to contribute to an underestimation of discard mortality. However, experimental holding of fish can also contribute to mortality via stress and unsuitable holding conditions, leading to overestimation (Portz *et al.*, 2006). While proper experimental controls were not available for our study, the short hauls with minimal air exposure (<5 min) indicate that the tank induced mortality was minimal or nil.

5.5.2 Other lessons from tank-based experiments

Tank-based experiments are also useful for determining the potential of mitigation options for reducing discard mortality. Tank-based experiments can highlight the prospects of focusing on biological, environmental or gear measures applied within a specific fishery. Van Beek *et al.* (1990) provided a first tank-based evaluation of the main factors contributing to discard mortality in beam trawl fisheries. Haul duration and handling were detected as important technical drivers. These

factors were therefore kept constant in this study, enabling the investigation of other technical factors, notably catch weight. Technical and biological factors were of primary interest, because gear modifications can potentially increase survival probabilities without changes in profitability, e.g. through selectivity changes or reduction in catch weight (e.g. Revill *et al.*, 2005; Enever *et al.*, 2010). Fishing trips were conducted during winter to early spring to reduce temperature variability between trips. As water layers in the southern North Sea are typically tidally mixed (i.e. no thermocline), little to no differences were expected in temperature and salinity between the sea surface and bottom (Holligan *et al.*, 1989). Other environmental conditions (e.g. salinity, depth) were not controlled for, as selection of fishing grounds was based on the advice of local fishermen. Also, the spawning condition of the fish was not sampled, as skates probably spawn during the summer months (Walker *et al.*, 1997), and soles in mid-April in the Thames estuary and the Belgian coast (Fincham *et al.*, 2013). Depth, SST, air temperature and salinity were monitored and included in the modelling procedure, because they varied considerably between fishing trips (Table 5.1). Fishing trip was also included to account for unidentified environmental and/or biological factors.

Statistically significant drivers of sole survival were catch weight, fish length, depth and injuries, but their explanatory power was limited, especially for depth. The overall explanatory power of the sole model was low (<20 %); however total variability in the sole survivorship function was also low, as indicated by the limited variability of sole survival within and between injury classes in Figure 5.2. This is in concordance to other discard survival studies, which could not detect any significant factors associated with sole survival other than length and injuries (Van Beek *et al.* 1990; Revill *et al.*, 2013). In contrast, water temperature and condition of the fish have been indicated as significant drivers for plaice survival in beam trawl fisheries (Van Beek *et al.*, 1990; Revill *et al.*, 2013). This is also in contrast to skates' survival in this study, which varied considerably (Figure 5.2), and was significantly driven by their length and injuries, explaining 69.0 % of the model variability. In contrast to Enever *et al.* (2008; 2010), catch weight was not selected for skates. Catch weight increases compression in the codend, which could affect fish condition. The mean codend weight (+/-S.E.) in this study was 133.6 (+/-39.1) kg for hauls in which skates could be retrieved, whereas Enever *et al.* (2008; 2010) reported mean weights up to 253 (+/-30) kg, suggesting that there might be somewhat of a threshold for this effect (>200 kg). Mandelman and Farrington (2007) support this hypothesis for another elasmobranch, *Squalus acanthias*, where especially catch weights > 200 kg yielded rapid elevations in short-term mortality. The significance of a threshold catch weight for survival is however speculative, as survival inevitably also depends on interacting effects with catch composition and behaviour of the codend in the water column. This nevertheless points out that flatfish are more susceptible to suffocation in the codend. This could be due to an increased risk of pressure on the operculum, as

suggested by Davis (2002) when comparing flatfish and roundfish. The investigated factors highlight that especially length is key to improved discard survival across fish species (Benoît *et al.* 2013). This implies a reinforcement of the existing need for increased size selectivity not only for discard reduction, but also for increased discard survival.

The number of potential factors affecting survival and the number of species examined was modest in this study due to constraints in our capacity to hold organisms for observation. Using quantified proxies for survival instead of full survival studies can ease this constraint (e.g. Benoît *et al.*, 2013; Davis, 2010). Physical injuries are easily quantified for both invertebrates and fish, and constitute one such proxy that was found to be effective for invertebrates, plaice and skates. However they were not a useful mortality proxy for sole and cod. Physical injuries can be assessed in an objective way by either presence or absence or by evaluating what percentage of the body had bruises, among others (Table 5.2, Table 5.3). The CDI was constructed to this end on the basis of multiple physical injuries since these can have a high explanatory power in predicting discard survival (Pálsson *et al.*, 2003). Each of the different CDI classes had equal weight in the total CDI score. The class “broken spine” can be classified as a lethal damage by itself and might have higher weight. However, we were not able to test this as there were no broken spines except for two isolated cases (cod and pouting). Here we have attempted to limit the subjectivity in determining a survival proxy by establishing quantitative rules for the CDI, thereby avoiding the need to address subjectivity posthoc during analysis (e.g. Benoît *et al.*, 2010). While physical injuries are quick and easy to measure and do not further complicate the already complex experimental designs, they do not fully predict survival as visible injuries are unrelated to reflex impairment, which is also a strong independent predictor (Davis and Ottmara, 2006; Stoner, 2012). Injury-based proxies that also incorporated the notion of fish “liveliness” were hence more effective in predicting post-capture mortality of sole (Van Beek *et al.*, 1990). An index based on reflexes, Reflex Action Mortality Predictor (RAMP), also indicated a good relationship with the mortality of cod (Humborstad *et al.*, 2009). Accounting for invisible, internal damage is hence indispensable for the development of a good proxy. Assessing liveliness though can be subjected to observer subjectivity (Benoît *et al.*, 2010), and constructing the RAMP can only be through assay validation and fish stressor experiments under laboratory conditions (Davis, 2010; Humborstad *et al.*, 2009). These additional constraints are an essential part of the complex, experimental design to study biological, environmental and gear-related factors of variability in discard survival. Yet, another proxy was recently developed, i.e. the time-to-mortality (TM) when a fish is exposed to air. This proxy indirectly includes both external and internal injuries. It provides an index of survival potential that is comparable to that obtained from more involved holding studies,

and can also be used to investigate the role of factors believed to affect discard mortality (Benoît *et al.*, 2013).

The constraints outlined above stress that interpretation of survival estimates from tank-based experiments should preferably be relative across taxa, and/or across other drivers, rather than absolute. A primary merit of tank-based experiments is hence identification of focal drivers for survival and consequent development of mitigating measures. They cannot be readily used in stock assessments, for instance. This would require an additional validation step, which could for instance be achieved with tagging experiments (Yergey *et al.*, 2012). Fisheries managers should be aware of the constraints of this and similar studies when considering measures such as a discard ban. Decisions on whether to impose a fishing ban based on the survival potential of particular discards need to acknowledge the lack of certainty of the estimates and need to be based on studies that have followed the best practices discussed here and by others (e.g., Davis 2002).

5.6 Acknowledgments

We are indebted to the crew of RV 'Belgica' for logistics during sampling, the Zoo Antwerpen for the holding tanks, and Miriam Levenson for linguistic comments. Ship time was provided by the Belgian Science Policy Office (BELSPO). We also thank ILVO's Fishing Gear Technology group of, in particular Fernand Delanghe and Kevin Vanhalst, for their support and advice. We thank Andy Revill and an anonymous referee for their valuable comments which improved the quality of this paper considerably.

5.7 Appendix

Table A5.1. Survival estimates of invertebrate discards in beam trawl fishery, based on holding tank experiments (≥ 48 h). The number of estimates is limited for a 4 m beam trawl with chain mat and haul duration >60 min, both for number of individuals and species. Number of individuals indicated in parentheses.

Species	Haul duration (<30 min)			Haul duration (60-150 min)		
	4 m chain mat ¹	4 m tickler chain ²	12 m tickler chain ³	4 m chain mat ²	4 m tickler chain ²	12 m tickler chain ⁴
Annelida (Polychaeta)						
<i>Aphrodita aculeata</i>	0.91 (125)		0.98 (653)	1 (15)		0.86 (248)
Mollusca (Bivalvia, Cephalopoda, Gastropoda)						
<i>Aequipecten opercularis</i>	0.97 (60)					
<i>Arctica sp.</i>			0.09 (130)			0.1 (1480)
<i>Chlamys sp.</i>						0.98 (53)
<i>Pecten maximus</i>	1 (65)					
<i>Spisula elliptica</i>						
<i>Spisula subtruncata</i>		0.59 (439)			0.68 (360)	
<i>Eledone cirrhosa</i>	0.88 (25)					
<i>Buccinum undatum</i>	1 (37)		0.4 (96)*			0.96 (171)
<i>Neptunia antiqua</i>	1 (35)					
<i>Euspira catena</i>			1 (10)			
Echinodermata (Asteroidea, Ophiuroidea, Echinoidea)						
<i>Asterias sp.</i>	0.99 (126)		0.93 (414)	1 (62)	0.96 (200)	
<i>Astropecten sp.</i>	1 (17)		0.93 (771)		0.93 (88)	0.91 (660)
<i>Crossaster papposus</i>	0.92 (24)					
<i>Luidia sarsi</i>			0.98 (246)			
<i>Ophiura sp.</i>		0.91 (85)	0.6 (133)	1 (59)	0.91 (153)	0.88 (520)
<i>Psammechinus miliaris</i>	0.38 (100)					
Arthropoda (Crustacea)						
<i>Cancer pagurus</i>			0.58 (12)		0.66 (53)	0.14 (21)
<i>Corystes cassivelaunus</i>			0.48 (872)		0.5 (14)	0.34 (1667)
<i>Crangon sp.</i>		0.92 (106)				
<i>Liocarcinus sp.</i>	0.58 (120)	0.84 (88)	0.62 (803)		0.61 (150)	0.47 (275)
<i>Macropodia rostrata</i>	0.74 (23)					
<i>Nephrops sp.</i>			0.83 (40)			0.58 (45)
<i>Pagurus sp.</i>	0.92 (169)		1 (30)	1 (23)	0.93 (27)	0.8 (244)
<i>Portunidae</i>				0.86 (99)		

*Only survival estimate of an observation period which is much longer than the others (37 days, Mensink *et al.*, 2000).

Table A5.1 (continued). Survival estimates of fish discards in different configurations of beam trawl fishery. All figures are based on survival experiment in holding tanks (>=48 h). Note that the number of estimates is limited for a 4 m beam trawl with chain mat (number of individuals) and for roundfish, elasmobranchs and non-commercial fish. No figures exist for a 12 m beam trawl with chain mat, except *Scyliorhinus canicula*. Number of individuals indicated in parentheses.

Species	Haul duration (<30 min)			Haul duration (60-150 min)				
	4 m chain mat ¹	4 m tickler chain ²	12 m tickler chain ³	4 m chain mat ²	4 m tickler chain ²	9 m chain mat ⁴	12 m chain mat ⁴	12 m tickler chain ⁵
Roundfish								
<i>Gadus morhua</i> ^a								
<i>Merlangius merlangus</i>			0 (42)					
<i>Trigla sp.</i>			0 (18)					
Elasmobranchs								
<i>Raja naevus</i>	0.59 (32)							
<i>Scyliorhinus canicula</i>	0.93 (42) ^b			0.98 (120)				
Flatfish								
<i>Solea solea</i>		0.87 (272)	0.47 (1202)	0.12 (104)	0.2 (275)	0.53 (114+50) ^c	0.76 (186+40) ^c	0.16 (581)
<i>Pleuronectes platessa</i>	0.39 (122)	0.94 (336)	0.34 (3568)	0.31 (55)	0.07 (425)	0.37 (515+40) ^c	0.47 (207+40) ^c	0.1 (2051)
							0.80 (256+40) ^c	
<i>Plathichthys flesus</i>		0.79 (14)		0.25 (8)	0.06 (341)			
<i>Limanda limanda</i>	0.23 (22)	0.82 (350)	0.52 (1063)	0.02 (166)	0.01 (3984)			
<i>Microstomus kitt</i>			1 (2)					
<i>Scophthalmus rhombus</i>					0.19 (31)			
<i>Scophthalmus maximus</i>					0.58 (50)			
Non-commercial fish								
<i>Agonus cataphractus</i>			1 (7)					
<i>Arnoglossus laterna</i>			0 (33)					
<i>Buglossidium luteum</i>		0.16 (44)	0.75 (155)					
<i>Callionymus lyra</i>	0.16 (115)	0 (19)	0.52 (23)					
<i>Pomatoschistus sp.</i>		0.95 (19)						
<i>Trachinus vipera</i>			0.94 (254)					

^a Lindeboom and de Groot (1998: 167) considered all gadoids dead within few minutes after being brought aboard

^b Survival estimate based on 12 m beam trawling with chain mat

^c Total mortality estimated were based on immediate and short-term mortality estimates (first and second number in parenthesis) and adjusted for control deaths. (Revill *et al.*, 2013)

¹ Kaiser and Spencer, 1995; ² de Groot and Lindeboom, 1994; Keegan, 2002; ³ Bergman *et al.*, 1990; de Groot and Lindeboom, 1994; Fonds, 1991; Keegan, 2002; mensink *et al.*, 2000; Van Beek *et al.*, 1990; ⁴ Revill *et al.*, 2013; ⁵ de Groot and Lindeboom, 1994; Fonds, 1991; Fonds *et al.*, 1992; Van Beek *et al.*, 1990

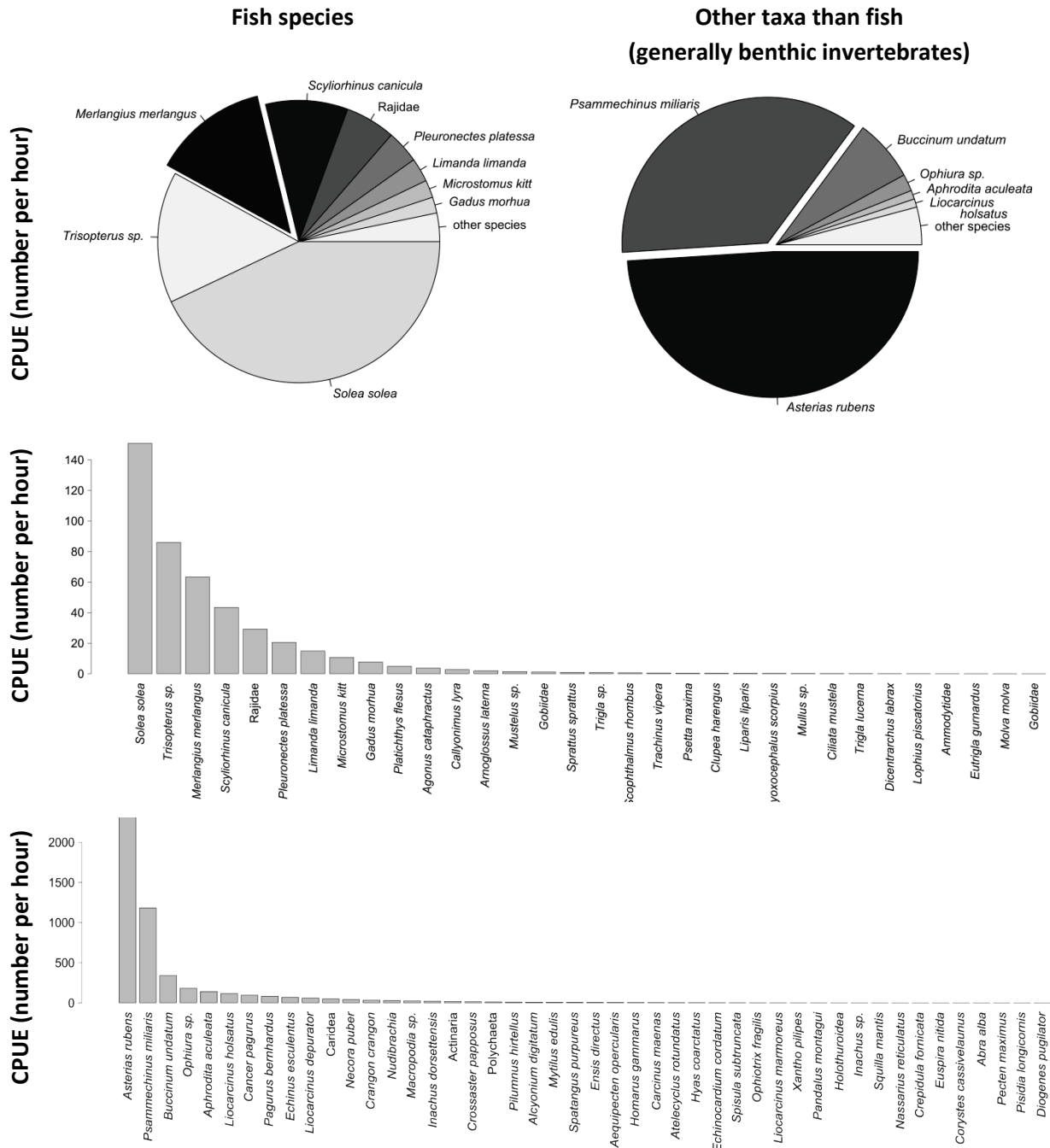


Figure A5.1. Mean number of individuals discarded by species in one hour beam trawling in the North Sea during the campaigns of the short-term survival study. CPUE: catch per unit effort in numbers. Upper panel: pie charts of fish species (left) and other taxa than fish (right). Middle panel: bar plot of fish species. Lower panel: bar plot of other taxa than fish.

6 Towards discard survival at fleet level

Life is a constant process of dying. – A. Schopenhauer

6.1 Introduction

Fisheries management requires that not only the discarded quantities are estimated at fleet level (), but also requested ‘scientific evidence’ at fleet level on the survival rates of species under the landing obligation (EU, 2013a). Current estimates of short-term survival vary between virtually zero and >90%. This variation may be due to uncertainties resulting from various methodologies (ICES, 2014c) or from the variability in survival resulting from a wide range of factors that significantly affect the stress levels and damage induced to organisms during the fishing and discarding process.

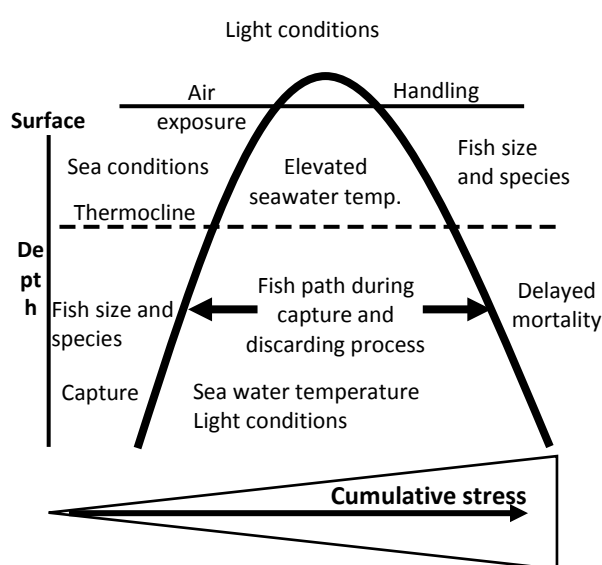


Figure 6.1 The capture process (curved line) illustrates the number of interacting factors that determine discard survival After Davis (2002).

The factors can be categorized according to biological (taxa, physiology, size, etc.), technical (haul duration, gear modifications, gear weight and design, etc.) and environmental (fishing depth, sea state, visibility, sediment, etc.) conditions. This vast suite of factors and their antagonistic or synergistic effects prevent accurate discard survival estimates at fleet level. Estimates are seemingly contradictory in many instances, hampering scientific assessments on the implications of survival for any stock (Figure 6.1; STECF, 2014b; see Introduction).

The conducted studies that have estimated discard survival in beam trawl fisheries are also limited due to the complex logistics of survival studies (Table A5.1; ICES, 2015e). Most of them (if not all) have focused on the short-term survival using holding facilities in on-board tanks (Chapter 5; Revill *et al.*, 2013; on-going survival projects; ICES, 2014c; 2015f) or underwater cages (Uhlmann *et al.*, 2014a; van Marlen *et al.*, 2013). The complexity of studying the survival of discards implies that a limited

number of species and/or individuals can be investigated and hence also a modest numbers of influential factors. Survival proxies may overcome this issue. Proxies do not directly measure survival, but are closely correlated with it and should have a 'reasonable' power to predict discard survival (ICES, 2014c). Proxies are easier and more rapid to estimate and can be obtained at a low cost, increasing the range of conditions by which survival can be examined (Davis, 2010). If proxies are validated in *in situ* situations, they can be used to rapidly generate discard survival estimates for different seasons, regions and fishing practices (Raby *et al.*, 2012; ICES, 2014c).

Physical injuries have limited capacities to predict survival (Chapter 5), but may be complemented by proxies that reflect stress such as changes linked to metabolic activity and thermal physiology (Skomal & Mandelman, 2012). The Reflex Action Mortality Predictor (RAMP) is an example of a survival proxy which originally used exclusively criteria that reflect stress such as impairment due to low oxygen levels (Davis, 2005; 2007; 2010; Davis & Ottmar, 2006). RAMP is based on behavioural reflexes, which are defined as stereotyped movements induced by a peripheral stimulus. Reflexes are impaired when organisms are stressed and can be correlated with reduced growth and changes in behaviour such as increased predator susceptibility (Olla & Davis, 1989). Reflex impairment has also been consistently correlated with survival in several studies (Barkley & Cadrin, 2012; Davis, 2010; Gallagher *et al.*, 2014; Hammond *et al.*, 2013; Humborstad *et al.*, 2009; ICES, 2014c; McArley & Herbert, 2014; Raby *et al.*, 2012; Stoner, 2012a; 2012b).

The RAMP-methodology meets the required criteria for the development of a proxy that can be quantified in a range of conditions, used to predict mortality and may therefor assists in the estimation of the discard survival at fleet level (ICES, 2014c) in beam trawl fisheries, and may be suitably complemented by the assessment of physical injuries, thereby improving its predictive power in estimating discard survival by proxies (Davis, 2005; Nguyen *et al.* 2014; Yochum *et al.*, 2015; Chapter 5).

RAMP is a method that involves checking for the presence or absence of natural animal reflexes to generate a behaviour (RAMP) score which is then used to predict discard survival (Davis, 2007; 2010). Well-defined external stimuli including light, gravity, sound and touch trigger innate reflexes to which irrevocably a reflex is shown by an unstressed, healthy animal. Specimens that experienced the fishing process may be stressed and injured and their responsiveness to such stimuli impaired. Reflex impairment is scored '1' when no, weak or doubtful responses can be observed, while unimpaired reflexes (clear, strong, obvious responses) are scored zero. Reflexes are considered impaired in case of doubt. The responses are scored by presence and absence to avoid variation due to biological factors such as size and gender and to minimise bias from subjective interpretations of the response

criteria. A series of reflexes is tested and the impairment scores summed to give a measure of impairment relative to the maximum possible score, i.e. the total number of tested reflexes. This proportion of reflex impairment is termed the 'RAMP-score' and varies between zero and one (no and full impairment respectively). The development of RAMP as a proxy to discard survival estimates is summarized in four steps:

1. Select species-specific candidate reflexes

The selection of candidate reflexes is crucial, as they need to be innate, involuntary responses that are present in unimpaired individuals and independent of size, maturity or other sources of within-species variability related to volitional behaviour (e.g. hunger). Flatfish respond differently to stimuli than roundfish or benthic invertebrates, or have a morphology and/or behaviour implying that not all reflexes are suitable for any one species (Barkley & Cadrin, 2012; Hammond *et al.*, 2013; Humborstad *et al.*, 2009). A total of 34 reflexes have been reported in scientific literature (Depestele *et al.*, 2014a. ICES, 2014c), but they have not been described in sufficient detail to repeat them consistently. The first step requires that candidate reflexes are selected and tested on a limited number of unstressed specimens to evaluate their species-specific applicability and to be able to describe them in detail to increase the repeatability of the tests in the protocol.

2. Test and select a final set of reflexes in reflex calibration tests

The suite of species-specific candidate reflexes from the previous step need to be standardized and tested for consistency in a sufficient number of unimpaired organisms (~20 individuals, Davis, pers. comm.). Unimpaired individuals may be sourced from aquaculture or from 'benign' wild-capture (ICES, 2014c). Hatchery reared or domesticated fish should be avoided as their stress responses or reaction to stimuli may differ from wild fish. Wild-captured fish have the advantage of being representative of the actual population, but the capturing methods may confound their responses. Capturing fish using 'benign' fishing methods is challenging, but may be approximated using traps, pots or via mild practices such as short hauls (Van Beek *et al.*, 1990; Chapter 5). The quality of the reflex calibration tests is assessed by evaluating the physical injuries of tested specimens and their short-term survival probability. Injuries should be minimal, while survival should be maximal. The reflex calibration tests are preferably conducted on specimens of varying conditions (lengths, maturity stages, etc.), and in conditions that minimize external variation (e.g. limited number of observers).

3. Validate species- and fishery-specific reflexes in relation to a stress gradient

The sensitivity of the reflexes to the variability in stress experienced in the fishery is tested in the validation step. The gradient of stress to which fish should be exposed requires a clear identification of the relevant stressors, as well as an appropriate design of stress experiments that will result in a gradient from fully impaired to unimpaired individuals. The development of new reflexes may be required when reflex responses do not cover the entire range between fully and unimpaired individuals. Step 2 and 3 of the RAMP framework should thus be applied in an iterative manner to develop species-specific reflexes that are sensitive to the fishery-specific stress gradient.

4. Model discard survival in function of the RAMP-score

Survival estimates should be estimated for individuals that were subjected to the stress gradient of the fishery as determined in step 3. The modelled relationship between reflex impairment (based on the RAMP-score) and survival may then be used to predict survival and evaluate fishery effects over a broader range of fishery conditions (Raby *et al.*, 2012). Survival estimates may be short-term survival estimates (based on holding facilities) or long-term survival (tagging studies) (ICES, 2014c; 2015e; 2015f).

RAMP is specifically developed for each species within a fishery as the impairment of reflex responses and physical injuries are fishery- and species-specific (Davis, 2010). As RAMP was developed in the U.S., it has not been applied to many species in European waters except for Atlantic cod (Humborstad *et al.*, 2009). This study initiated the development of RAMP-scores for plaice and sole in Belgian beam trawl fisheries (Depestele *et al.*, 2014a). The objectives were the identification of suitable candidate reflexes for plaice and sole and an investigation of the responses of unimpaired individuals with high survival probabilities.

6.2 Materials and methods

The objectives of this study focused on the first and second step of the RAMP framework, which were applied to sole and plaice in flatfish-directed beam trawl fisheries. An extensive list of candidate reflexes was evaluated for wild-captured specimens using individual interpretations of observers and group discussions. A set of candidate reflexes was finalized and tested in calibration tests. Unimpaired individuals were evaluated using on-board holding facilities and physical injuries.

6.2.1 Sea trial and selection of unimpaired fish

Experiments were conducted between 3-7 March 2014 aboard the Rv 'Belgica' in the southern North Sea (ICES Subarea IVc, between 51.25° and 52,25° N and 1.5° and 3.5° E). Fishing was conducted with a two 4 m beam trawls and mimicked commercial practices of the 'eurocutter' fishery (see Chapter 5 for a detailed description). A total of six short hauls were conducted for the development of the RAMP (Table 6.1 or <http://odnature.naturalsciences.be/belgica> for further details).

Table 6.1. Specifications of the hauls that were used to collect unimpaired sole and plaice.

Haul	Depth (m)	Towing direction*	Current speed (kn)	Towing speed (kn)	Wind force (Beaufort)	Wave height (m)
1	34-37	against	1.3	3.5-4	5	1
2	26-33	against	2	3.5-4	4	0.5
3	24-25	with	1.7	3.5-4	4	0.5
4	37	with	2.1	3.5-4	4	0.5
9	45-52	against	1	3.5-4	3	0.5
10	48-51	against	2	3.5-4	3	0.2

* towing direction is either against the current, with or perpendicular to the current

Hauls lasted <20 min to ensure high survival probabilities (Chapter 5). The catch was released on the deck. Sole and plaice were selected from the catch. Only vivid individuals were collected and directly collected in buckets filled with water to minimize air exposure. Fish were brought to the sorting table and extracted from the bucket to assess their reflexes. Fish were measured to the nearest cm below after assessing their reflexes. The reflex assessments were conducted by one observer to minimize observer bias. The descriptions of the reflexes, which preceded reflex assessments, were evaluated by a group of four observers which jointly interpreted the reflexes from literature. All tests were conducted on separate hauls and ended within 15 min to reduce the stress of holding fish on-board.

6.2.2 Candidate reflexes for sole and plaice

Ten candidate reflexes were *a priori* selected from successful reflex tests in flatfish survival experiments (Barkley & Cadrin, 2012; Yergey *et al.*, 2012) and advice by M. Davis.

First, the description of the ten reflex tests was interpreted, tested and fine-tuned using four specimens of plaice (Total Length (TL): 19, 26, 29, 29 cm) from the first haul (H1, Table 6.1) and four soles from the second haul (TL: 19, 24, 27, 39 cm) (H2). The various interpretations and manipulations led to detailed descriptions of the reflexes and the formulation of additional, species-specific reflexes.

Second, a selection of 16 reflexes was qualitatively evaluated by a group of four observers, based on manipulations and the detailed descriptions of the previous hauls (H1 and H2). The qualitative evaluation led to a further selection of ten reflexes which were formally tested using the fine-tuned reflex descriptions. The ten reflexes were tested on specimens retrieved from haul 3: three plaice (TL: 21, 22, 24 cm) and six soles (TL: 18, 20, 20, 24, 25, 27 cm) and haul 4: four plaice (TL: 23, 36, 38, 45 cm) and five soles (TL: 19, 22, 29, 29, 32 cm). The tests of these ten reflexes resulted in a final set of seven reflexes for the calibration tests for sole and plaice.

6.2.3 Reflex calibration tests

6.2.3.1 Protocol and assessment of reflexes

Seven candidate reflexes were tested on 22 individuals of sole with a mean TL (SD) of 26.3 (4.6) and plaice with a mean TL of 25.6 (5.1), caught in two hauls (H9 and H10, Table 6.1). The order of testing reflexes was *a priori* determined to avoid contamination of responses across individuals and to enable rapid assessment (reducing stress). Evading for instance was tested after the fish have been held in the observer's hands, after which the stabilising reaction was tested (Table 6.1). The tested reflexes for sole were (in order of testing): *righting* (R1), *vestibular-ocular response* (R2), *head* (R3), *evade* (R4), *stabilise* (R5), *mouth* (R6) and *tail grab* (R7). The tested reflexes for plaice were (in order of testing): *righting* (R1), *vestibular-ocular response* (R2), *evade* (R3), *stabilise* (R4), *operculum* (R5), *mouth* (R6) and *tail grab* (R7). The total length ranged between 16 and 32 cm for sole and 17 and 39 cm for plaice. RAMP-assessments lasted < 3 minutes by individual.

6.2.3.2 Quality assessment of the calibration tests

Physical injuries were assessed after the RAMP scoring using the modified Catch Damage Index (CDI, Table 5.3), which was presented relative to the maximum score (rCDI). Fish were also transferred to on-board holding facilities and monitored for 60 hours. Details on the holding facilities and further technicalities were described in Chapter 5, with the exception of the number of tested individuals by tank. One sole and one plaice were placed in a tank, which enabled matching the survival assessment with the RAMP and CDI scores and avoiding tagging.

6.3 Results

6.3.1 Candidate reflexes for sole and plaice

6.3.1.1 Description of candidate reflexes

A detailed description of the interpretation of the literature-based reflexes and the newly developed reflexes resulted in a total of 19 detailed descriptions of reflexes (Table 6.2; Table 6.3 and Figure 6.2).

Table 6.2. Fine-tuned description of ten candidate reflexes that were *a priori* selected from successful reflex tests in flatfish survival experiments and advice by M. Davis.

Reflex	Stimulus inducing the reflex	Reflex response in unimpaired individuals
1. <i>Righting</i>	Turn fish upside down (belly facing the ceiling) in a water column > the total length of the tested individual.	Fish returns to normal orientation within 5 seconds.
2. <i>Body flex 2</i>	Place fish in air and shift it horizontally for 5 cm on a flat and smooth surface (typically a measuring board).	Body flexes multiple times within 5 seconds, i.e. the fish starts ' <i>flapping</i> ', a typical behaviour observed when measuring sole and/or plaice.
3. <i>Head complex</i>	Look at the head complex, especially gills and mouth, when the fish is in water	Regular pattern of ventilation with jaw and operculum is clear within 5 sec.
4. <i>Vestibular-ocular response (VOR)</i>	Hold the fish horizontally in air (belly facing downwards) and look at the eyes. Rotate the fish along its longitudinal axis and look at the eyes.	Eyes stay within a horizontal plane. Eyes of impaired individuals follow the rotations and are not actively moved. Eyes may also retract into the body, especially for sole.
5. <i>Body flex 1</i>	Place the full hand without gloves along the longitudinal axis onto the fish for 5 seconds (in water).	Active body movements to escape or resistance to being covered.
6. <i>Fin control</i>	Hold the fish in water. Gently brush the fins along the longitudinal axis with a blunt object and continue for 5 sec.	Fish move their fins along the longitudinal axis as if they were to avoid tickling.
7. <i>Tail grab</i>	Hold the fish in water. Gently grab the tail between two fingers with the intention to hold it.	Active body movements to escape the tail grab.
8. <i>Operculum</i>	Hold the fish in water. Gently open the operculum with a blunt object. The reaction should be clear after a maximum of 5 attempts.	Clear resistance to open the operculum (sole) or a clear reaction to close it when the object is removed or slight resistance to open the mouth (plaice)
9. <i>Mouth</i>	Hold the fish in water. Gently open the mouth with a blunt object. The reaction should be clear after a maximum of 5 attempts.	Clear resistance to open the mouth (sole) or a clear reaction to close it when the object is removed or slight resistance to open the mouth (plaice)
10. <i>Evade</i>	Test this reflex after a reflex test in air. Fish is gently released at the water surface.	Fish actively swim away using clear muscle movement and not just drifting in water.

Table 6.3. Formulation of additional reflexes for plaice and sole.

Reflex	Stimulus inducing the reflex	Reflex response in unimpaired individuals
11. <i>Head</i>	Hold the fish for 5 sec horizontally in air (belly facing downwards) between thumb and fingers. The hand only touches the head complex without pressing severely.	Active upward curling of the body at least once and reaching the horizontal plane.
12. <i>Fin lift</i>	Wait until the fish is (actively or passively) put onto the bottom of a box with water. Lift the dorsal and anal fins with a blunt object.	Fish move their fins along the longitudinal axis as if they were to avoid the disturbance and attempted to dig into the sediment or keep position on the bottom of the box.
13. <i>Headbang</i>	Hold the fish in water. Gently lift its head with a blunt object for ~1 cm.	Fish lift head (and/or tail) as if they were to dig into the sediment or to avoid the disturbance.
14. <i>Wave</i>	Fish are freely in the water on the bottom of the box. Lift the pectoral fin with a blunt object.	Fish actively move their fin and simulate a waving pattern instead of returning to its original position along the longitudinal axis.
15. <i>Stabilise</i>	A fish is released in a box with water with or without sediment (e.g. after testing R10, <i>evade</i>). Look at the fins of the fish for maximum 5 sec after they have reached the bottom of the box. Ship movement may alter this reflex.	Fish move their fins along the longitudinal axis (dorsal, anal fin) as if they were to dig into the sediment.
16. <i>Gag</i>	Hold the fish in air and gently insert a blunt object into the mouth at maximum 1 cm. Release the object. This test may be performed once after testing R9, mouth.	Fish move gills and mouth and spit out the object.
17. <i>Cover</i> (not tested)	Gently put the fish into a box filled with sea water and sandy sediment. Wait until the fish have reached the sediment and observe their reaction for maximum 5 sec.	Fish attempt to dig into the sandy sediment.
18. <i>Tail push</i> (not tested)	Hold the fish in water. Push hard on the tail with the thumb.	Fish immediately respond by active body movements related to the disturbance.
19. <i>Darkness</i> (not tested)	Place the fish in a box with water. Cover part of the box to darken it and enlight the other half of the box. Release the fish in the light part and observe its reaction within 5 sec.	Fish actively move towards the darkened part of the box within 5 sec.

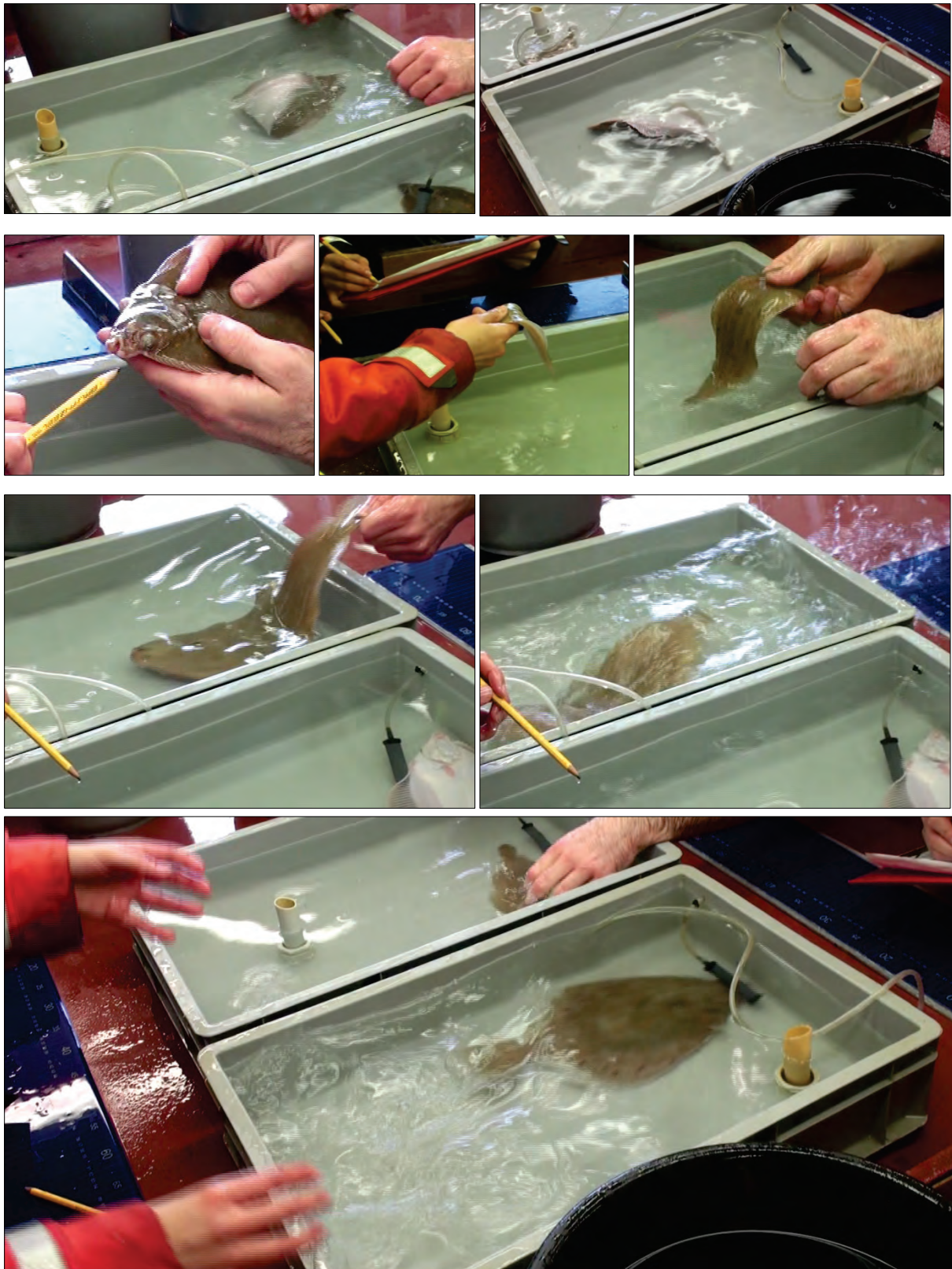


Figure 6.2. Illustration of a selection of candidate reflexes for sole and plaice (from left to right and top to bottom): R1 (*righting*) for sole, R1 for plaice, R4 (VOR) for plaice, R11 (*head*) for plaice, R11 for sole, R10 (*evade*) stimulus and R10 response for sole, R10 (*evade*) response for plaice. Details on the reflexes are described in Table 6.2 and Table 6.3.

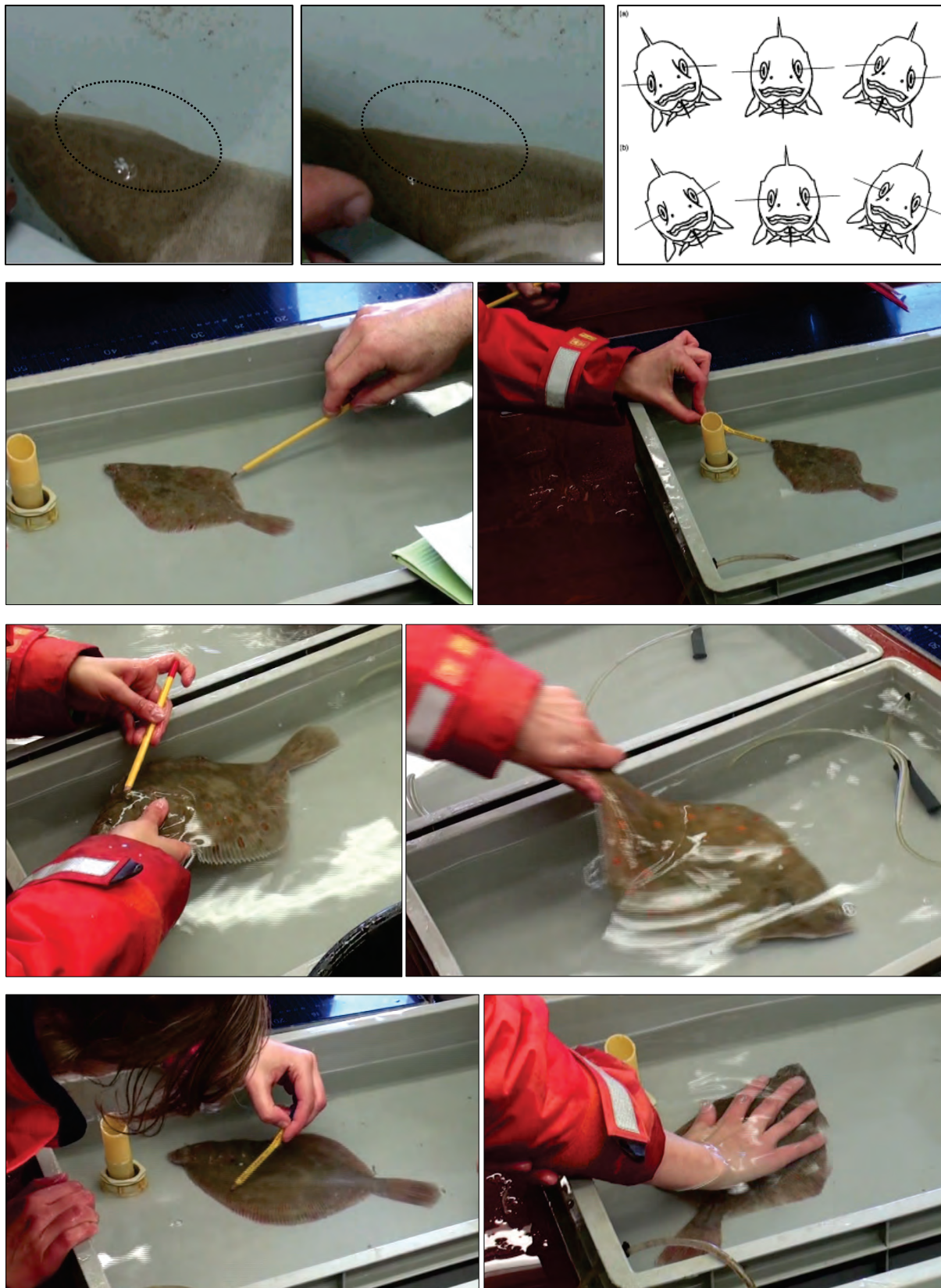


Figure 6.2 (continued). Illustration of a selection of candidate reflexes for sole and plaice (from left to right and top to bottom): R15 (*stabilise*) for sole (two pictures), R4 (VOR: schematic from Kestin *et al.*, 2002), R12 (*fin lift*): if stabilisation of the fish on the bottom is not apparent, the fins were lifted. R9 (*mouth*), R8 (*operculum*) R8 should be tested under water in contrast to what the picture shows, R7 (*tail grab*), R6 (*fin control*) and R5 (*body flex 1*). Details on the reflexes are described in Table 6.2 and Table 6.3.

6.3.1.2 Selection of candidate reflexes for the reflex calibration tests

Ten reflexes were selected as potential candidates for the reflex calibration tests for sole and plaice: *righting*, *body flex 2*, *Vestibular-ocular response*, *tail grab*, *operculum*, *mouth*, *evade*, *head*, *fin lift*, *stabilise* and one additional reflex: *gag* (see Table 6.2 and Table 6.3 for detailed descriptions: R1, R2, R4, R7, R8, R9, R10, R11, R12, R15 and R16).

The reflex calibration tests for sole indicated five reflexes with consistent responses (R): *righting* (R1), *tail grab* (R4), *evade* (R8), *stabilise* (R9) and *head* (R10). Two additional responses were selected, although they did not show a fully consistent response: *vestibular-ocular response* (R3) and *mouth* (R6). Three reflexes were disregarded, as doubts and contradiction were dominated the tests: *body flex 2* (R2), *operculum* (R5) and *gag* (R7).

The reflex calibration tests for plaice yielded consistent results for five responses: *vestibular-ocular response* (R3), *operculum* (R5), *gag* (R7), *evade* (R8) and *stabilise* (R9). Three reflexes were not fully consistent, but were nevertheless put forward to be experimentally tested in the reflex calibration tests: *righting* (R1), *tail grab* (R4) and *mouth* (R6). Seven reflexes were selected for the reflex calibration tests, eliminating *gag* (R7) due to a lack of consensus among observers on appropriate application and description.

6.3.2 Reflex calibration tests

6.3.2.1 Assessment of reflexes

The following reflexes were tested for both sole and plaice: *righting* (R1), VOR (R3), *evade* (R8), *stabilise* (R15), *mouth* (R6) and *tail grab* (R4). The reflex '*head*' (R3) was also tested for sole, but not for plaice. The reflex '*operculum*' (R5) was not selected for sole, but it was for plaice. All of the reflexes were unimpaired for six individuals of sole and ten of plaice (Table 6.4 and Table 6.5). The mean RAMP-score for sole was 0.18 (SD: 0.16) and 0.16 (SD: 0.20) for plaice, when seven reflexes were considered. Three individuals of sole and plaice had a RAMP-score ≥ 0.43 , i.e. several reflexes were impaired. The RAMP-scores were not correlated with length for sole ($r=0.12$, p -value=0.58) or for plaice ($r=0.36$, p -value = 0.10).

The *tail grab* was unimpaired among all tested soles, while VOR and the *operculum* were fully consistent for plaice. Responses to the other reflexes were <6 times absent for sole, except for *evade*. *Evade* was not present in 12 out of 22 tests. The responses of plaice showed a higher consistency (impaired in <6 tests), except for *righting*. Plaice did not turn to its belly in 9 out of 22 tests.

6.3.2.2 Quality assessment of the calibration tests

Nearly 60% of the sole specimens did not have any physical injuries, while the number of plaice without external damage was limited to 27 %. The mean rCDI (SD) was 0.04 (0.06) for sole and 0.10 (0.08) for plaice. The physical injuries of sole and plaice were low, although not absent in all individuals. There was no correlation between the RAMP and the rCDI-scores for sole ($r=-0.19$, p -value=0.39) or for plaice ($r=0.26$, p -value =0.24), although the number of plaice without any physical injury was seemingly higher at low RAMP-scores (Table 6.6 and Table 6.7), confirming the contribution of injuries to its impairment (Chapter 5). Physical injuries were generally low (rCDI < 0.25), while the RAMP-score was high for some individuals (RAMP > 0.25) (Figure 6.3).

None of the individuals of plaice died during the observation period, indicating a high survival probability for all tested individuals. The survival probability of sole was also high, although one of the specimens died at 48 hours of observation. The dead individual was 16 cm and had a RAMP-score of zero and rCDI of 0.14. The reflex calibration tests were hence conducted on individuals with high survival probabilities, but which were, however, not fully unimpaired.

Table 6.4. RAMP-score of sole in the reflex calibration tests. The tested reflexes were: (1) *righting*, (2) *vestibular-ocular response*, (3) *head*, (4) *evade*, (5) *stabilise*, (6) *mouth* and (7) *tail grab*. Unimpaired reflexes were scored zero.

L	1	2	3	4	5	6	7	RAMP
28	0	0	0	0	0	0	0	0
24	0	0	0	0	0	0	0	0
27	0	0	0	0	-	0	0	0
32	0	0	0	0	0	0	0	0
22	0	0	0	0	0	0	0	0
26	0	0	0	0	0	0	0	0
16	0	0	0	0	0	0	0	0
22	0	0	0	1	0	0	0	0.14
30	0	0	0	1	0	0	0	0.14
22	1	0	0	0	0	0	0	0.14
24	0	0	0	1	0	0	0	0.14
37	0	0	0	1	0	0	0	0.14
31	0	0	0	1	0	0	0	0.14
28	0	0	0	1	0	0	0	0.14
24	1	1	0	0	0	0	0	0.29
20	1	0	1	0	0	0	0	0.29
24	0	0	1	1	0	0	0	0.29
28	1	0	0	1	0	0	0	0.29
30	0	0	1	1	0	0	0	0.29
27	0	0	1	1	1	0	0	0.43
30	0	0	1	1	0	1	0	0.43
27	1	1	1	1	0	0	0	0.57

Table 6.5. RAMP-scores of plaice in the reflex calibration tests. The tested reflexes were: (1) *righting*, (2) *vestibular-ocular response*, (3) *evade*, (4) *stabilise*, (5) *operculum*, (6) *mouth* and (7) *tail grab*. Unimpaired reflexes were scored zero.

L	1	2	3	4	5	6	7	RAMP
34	0	0	0	0	0	0	0	0
26	0	0	0	0	0	0	0	0
21	0	0	0	0	0	0	0	0
25	0	0	0	0	0	0	0	0
26	0	0	0	0	0	0	0	0
21	0	0	0	0	0	0	0	0
22	0	0	0	0	0	0	0	0
17	0	0	0	0	0	0	0	0
23	0	0	0	0	0	0	0	0
23	0	0	0	0	0	0	0	0
28	1	0	0	0	0	0	0	0.14
30	0	0	0	0	0	0	1	0.14
26	0	0	0	0	0	1	0	0.14
20	1	0	0	0	0	0	0	0.14
24	1	0	0	0	0	0	0	0.14
24	1	0	0	0	0	0	0	0.14
28	1	0	0	0	0	0	1	0.29
32	1	0	0	0	0	0	1	0.29
20	0	0	0	1	0	0	1	0.29
29	1	0	0	1	0	0	1	0.43
27	1	0	1	1	0	1	0	0.57
39	1	0	1	1	0	1	1	0.71

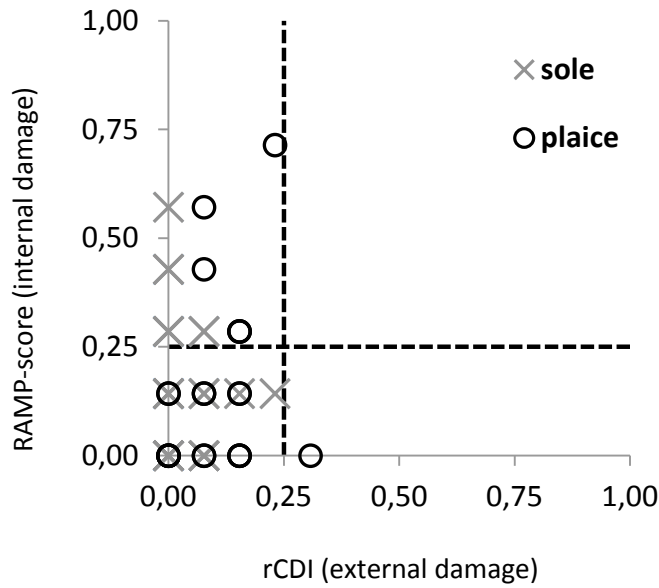


Figure 6.3. The RAMP and rCDI scores for individuals of sole and plaice in the reflex calibration tests. Injuries (rCDI) were low (<0.25), but RAMP-scores were high (>0.25) for a few specimen.

Table 6.6. CDI scores for sole in order of increasing RAMP-score. Numbers relate to the CDI categories: 1: gear-related damages, 2: skin-abrasion, 3: bruises for (a) *head*, (b) *tail* and (c) *body*, 4: pressure injuries, 5: fin and tail damage.

L	1	2	3a	3b	3c	4	5	rCDI
28	0	0	0	0	0	0	1	0.14
24	0	0	0	0	0	0	0	0
27	0	0	0	0	0	0	0	0
32	0	0	0	1	0	0	0	0.14
22	0	0	0	0	0	0	0	0
26	0	0	0	0	0	0	0	0
16	0	0	1	0	0	0	0	0.14
22	0	1	0	0	1	0	1	0.43
30	0	0	0	0	0	0	0	0
22	0	0	1	0	0	0	0	0.14
24	0	0	0	1	0	0	0	0.14
37	1	0	0	0	1	0	0	0.00
31	0	0	0	0	0	0	0	0
28	0	0	0	0	0	0	0	0
24	0	0	0	0	0	0	0	0
20	0	0	0	0	0	0	0	0
24	0	0	0	0	0	0	0	0
28	0	0	1	0	0	0	0	0.14
30	0	1	0	0	0	0	0	0.14
27	0	0	0	0	0	0	0	0
30	0	0	0	0	0	0	0	0
27	0	0	0	0	0	0	0	0

Table 6.7. CDI scores for plaice in order of increasing RAMP-score. Numbers relate to the CDI categories: 1: gear-related damages, 2: skin-abrasion, 3: bruises for (a) *head*, (b) *tail* and (c) *body*, 4: pressure injuries, 5: fin and tail damage.

L	1	2	3a	3b	3c	4	5	rCDI
34	0	0	2	0	0	0	0	0.29
26	0	0	0	0	0	0	0	0
21	0	0	0	0	0	0	0	0
25	0	1	1	0	0	0	0	0.29
26	0	0	0	1	0	0	0	0.14
21	0	0	0	0	0	0	0	0
22	0	1	2	1	0	0	0	0.57
17	0	0	0	0	0	0	0	0
23	0	1	0	0	0	0	0	0.14
23	0	1	0	0	0	0	1	0.29
28	0	0	0	0	0	0	0	0
30	0	0	1	0	0	0	0	0
26	0	0	1	0	0	0	0	0.14
20	0	0	0	0	0	0	0	0
24	0	1	0	1	0	0	0	0.29
24	0	0	0	1	1	0	0	0.29
28	0	1	0	0	0	0	1	0.29
32	0	1	1	0	0	0	0	0.29
20	0	1	1	0	0	0	0	0.29
29	0	0	0	0	0	0	1	0.14
27	0	0	0	1	0	0	0	0.14
39	0	1	1	1	0	0	0	0.43

6.4 Discussion

6.4.1 Identification and calibration of reflexes for sole and plaice

A total of seven reflexes were selected for sole and plaice after extensive discussion and testing of a >15 reflexes (Figure 6.4). The reflexes were clearly described by an action and response, but did not result in fully consistent responses during the reflex calibration tests of sole and plaice. The recommendation from this study is nevertheless to continue further development of the RAMP using all seven reflexes.

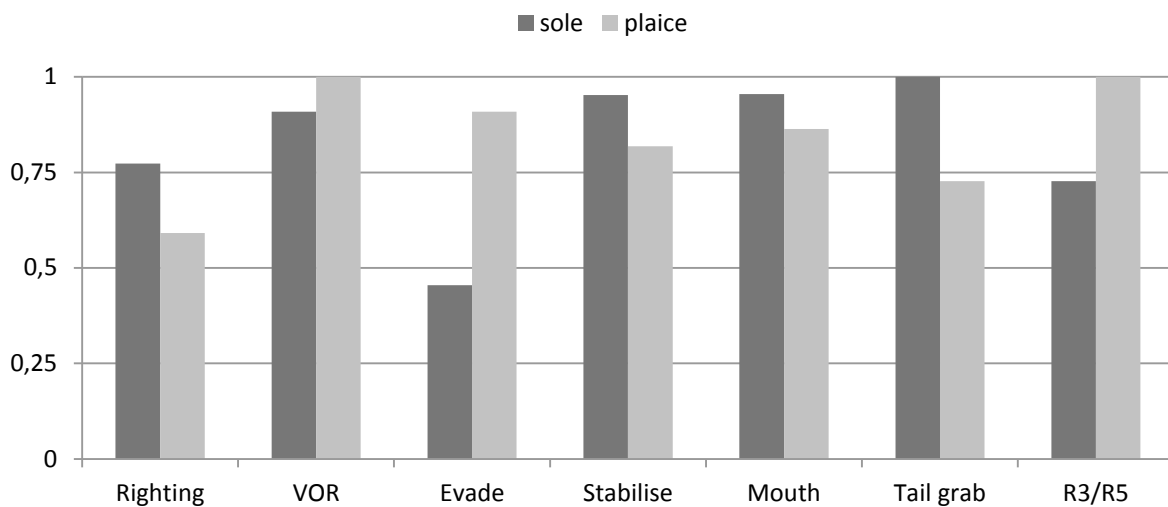


Figure 6.4. Proportion of unimpaired individuals of sole and plaice for the seven reflexes tested in the reflex calibration tests. R3 (*head*) was tested for sole, R5 (*operculum*) for plaice. A proportion of '1' implies that these reflexes should be selected for further RAMP-development.

The reflexes for sole that showed response for most individuals, were the *tail grab*, *mouth*, *stabilise* and VOR. The individuals of sole that were manipulated in air were in several instances difficult to handle as they curled around the hand of the observer. This behaviour is indicative of the *head* response and may in practice be evaluated from this type of observations. The *righting* reflex was observed regularly during manipulations besides the reflex calibration tests. In 5 out of the 22 tests, however, the specimens did not return to their natural position within 5 seconds except when they were stimulated. The stimulation is not part of the description of the reflex action, but may be indicative of the relevance of this reflex for assessing impairment. The *evade* reflex was rarely present for soles in the reflex calibration tests, which is in contrast to the preliminary tests that were conducted to fine-tune the detailed descriptions of the reflexes (chapter 6.3.1.2). All tested sole specimen (N=11) exhibited that response (Depestele *et al.*, 2014a), whereas 12 specimens of the formal reflex calibration tests drifted > 5 seconds before they turned to the natural position.

The most consistent reflexes of plaice were the *operculum* and the VOR, the eye roll. The eyes of plaice are differently placed than the ones of sole, and remained focused during rotation. While this response was obvious for the observer conducting the tests, it was less so for other observers. Inter-observer variation may especially complicate the evaluation of this reflex, although other reflexes may be susceptible to observer bias as well (Tuytens *et al.*, 2014). Bias between scores of several observers was avoided during this study, but may become relevant when RAMP scores are obtained by more than one scorer within the same study. The resistance to open the operculum was also clearly present in plaice, as were the reflexes *evade* and *tail grab*. *Mouth* induced a clear response, although required subtle manipulations and observations such as the slow closure of the mouth after opening. *Righting* of plaice was limited, which was probably due to the limited height of the water column in the holding tanks (<11 cm).

The lack of full consistency in most reflexes was likely due to the impairment of the fish. This observation was not captured by the low levels of physical injuries and not by the nearly 100% short-term survival of both plaice and sole, but it was noted from the high RAMP-scores of three plaice and three sole individuals. Kelle (1977) also showed that impaired plaice do not always die from their injuries. The sublethal impairment of the control individuals is further strengthened by expert judgment. The observers noted that 'vivid' individuals were more likely to show a response to the tested reflexes and that the organisms were more vivid after being submerged for some time in the on-board holding facilities. The reflex calibration tests were conducted directly after the sorting of the catch, which potentially compromised the vitality of the tested individuals by spinning them and other sublethal effects. The *tail grab*, for instance, required that the tail was actually grabbed and held between the fingers during the reflex calibration tests, whereas evaluating whether fish were dead or alive revealed that touching the tails was sufficient to induce a response.

The high RAMP-score of six individuals and the qualitative observations highlight that the acquisition of control individuals does not only require that their short-term survival is high and that their injuries are low, but also that sublethal effects are controlled for, as the low levels of physical injuries and the nearly 100% short-term survival did not show a statistical significant correlation with the RAMP-scores.

This study was intended to test reflexes in unimpaired individuals, but did not fully succeed to obtain unimpaired fish from their natural habitat. Unimpaired individuals may be obtained by using the same capture methodology (short hauls), but complemented with an accommodation period in holding facilities at sea or in the laboratory. Observers noted that vitality increased within 24 hours, but other studies illustrated that full recovery may take several weeks (Debuschere *et al.*, 2014;

Neat *et al.*, 2009). It can therefore be advised to repeat the reflex calibration tests for the same reflexes, but using individuals that have recovered after an accommodation period. The reflexes may then show full consistency in their responses. The results from this study may be further used in the validation step of the RAMP-development, representative for individuals that experienced 'light' levels of stress. Further development of the RAMP-methodology requires that RAMP and survival is estimated for individuals that experienced higher stress levels, e.g. by using individuals from longer hauls. The results of this study should therefore be interpreted as one of the necessary steps in an iterative framework to develop RAMP for sole and plaice in beam trawl fisheries.

6.4.2 *Survival of discards at fleet level*

6.4.2.1 *The development of proxies for discard survival*

Before the reflex-based RAMP method, survival of discards had been assessed and correlated with a semi-quantitative index of vitality in earlier survival studies. Vitality relates to the potential of an individual to survive the capture and discarding process, and is estimated from semi-quantitative vitality assessments (Van Beek *et al.*, 1990). High vitality implies unstressed and undamaged individuals. Low vitality indicates severe injuries or stress leading to high impairment.

Some proxies have attempted to define objective criteria to score vitality in a repeatable manner rather than a subjective evaluation as used in earlier studies (Enever *et al.*, 2008; Van Beek *et al.*, 1990). One of these proxies is based on the level of injuries (Catch Damage Index, CDI) and was particularly useful for benthic invertebrates, but also for Rajidae and plaice (Chapter 5). Injuries could not predict the mortality rate of soles and cod, and thereby highlight their limitations to also evaluate more subtle impairment in the short term, such as internal damages from endured stress. Time-to-mortality (TTM) is another, quantitative predictor that clearly discriminates between discard survival of several taxa, but it is typically related to hypoxia (Benoît *et al.*, 2013). TTM is the time that it takes for an organism to die while being out of the water and exposed to air. This indicator follows the projections that an increased handling time on deck clearly leads to reduced survival.

The Reflex Action Mortality Predictor (RAMP) is the proxy that was addressed in this study. It can easily be measured and quantified, and integrates a variety of stressors that are related to invisible, inner damage from fishing such as barotrauma, hypoxia, reduced metabolic activity, fatigue, etc. (Uhlmann *et al.*, 2015; Wilson *et al.*, 2014). RAMP was developed in the U.S. on the basis of behavioural reflexes, but should be complemented with criteria from physical injuries (Davis, 2005; Yochum *et al.*, 2015; Chapter 5). RAMP demonstrated its responsiveness to reduced vitality and correlated significantly with delayed mortality.

The development of RAMP, however, requires the application of a framework that is carefully addressed and fine-tuned by species and fishery (Yochum *et al.*, 2015). The calibration of relevant reflexes and their validation is an iterative process that cannot be created overnight (M. Davis, pers. comm.), as was illustrated by the results from this study. The experiments illustrated, however, that the sensitivity of RAMP to pick up signals was higher than that of physical injuries or short-term survival alone. It may therefore be concluded that the development of RAMP based on injuries and reflexes has a great potential as a proxy to predict discard survival, and that it advances to existing estimates by being quantitative and inclusive of both external as well as internal damage.

6.4.2.2 Using survival proxies in fisheries management

RAMP facilitates short-term survival estimates at fleet level

As RAMP-methodology can be easily taught to observers (Raby *et al.*, 2014b; Chapter 5), the national discard observer programmes may be used to collect a high number of RAMP-scores from various fishing trips, where are representative of commercially caught and discarded sole and plaice in the beam trawl fishery. As significant correlation between RAMP and discard survival is highly likely (Theunynck *et al.*, 2015), these fleet-wide RAMP-data may provide quantitative estimates of short-term discard survival over a variety of fishery conditions that are representative of the fishery at fleet level.

When short-term discard survival estimates based on the inferences of RAMP are combined with estimates of discarded quantities, they may be indicative of the implications of discard mortality for a single species stock. These estimates deliver quantitative data to better understand and discriminate between the contributions of various fisheries to discard-induced mortalities. In other words, the RAMP-survival relationship will not allow fisheries' scientists to decide what high or low survival rates are, but will assist in delivering the tools to fisheries' managers to indicate the order of magnitude by which specific fisheries contribute to discard-induced mortalities of a single-species stock.

RAMP facilitates the development of gear measures to increase discard survival

The development of RAMP may also facilitate the assessment of innovative gears that are not only developed to reduce discards (PARTIM III), but also to increase the survival rate of discards (Enever *et al.*, 2010). The discard survival of the endangered coho salmon (*Oncorhynchus kisutch*) in the beach seine fishery in Canada, for instance, was markedly lower than salmon survival in other Canadian fisheries (Raby *et al.*, 2014b). The causes of the low survival could not easily be examined by

monitoring all potential influential factors during survival studies. Instead, a significant relationship between RAMP and coho salmon survival was modelled (Raby *et al.*, 2012). The effect of the management measures on survival was then evaluated by the application of RAMP in a high number of commercial conditions. Raby *et al.* (2014b) concluded that a reduction in handling time significantly correlated to a decrease in reflex impairment, and thereby illustrated how existing RAMP-survival relationships may be used to facilitate the development of fishing practices which increase discard survival. The application was also facilitated by the ease of teaching the RAMP-methodology to observers. RAMP may thus also be taught to fishermen and empower them to develop gears that increase discard survival, which goes along with the incentive created by the landing obligation to have an exemption for fish with high discard survival.

Short-term discard survival as a proxy for *in situ*, long-term discard survival

The estimates of discard survival were generally estimated during a limited period of observation, typically lasting as long as the fishing trip or possibly extended by an observation period in the laboratory. They are typically conducted outside the natural habitat of the organisms, excluding the susceptibility to predation and diseases. These estimates of discard survival may be defined as short-term estimates, and are in itself also a proxy for 'real, *in situ*' discard survival in the long-term.

Long-term survival can be estimated by returning discarded organisms to their natural habitat and tracing them using mark-and-recapture studies or telemetry. The advantages of mark-and-recapture studies are their low costs and high sample sizes, but its main problem is the high uncertainty in estimating survival as the returned marks are generally low and marked individuals may die from natural causes (ICES, 2014c, Pollock & Pine, 2007). Telemetry is the remote registration of the behaviour, physiology and/or energetics of discarded organisms. It is an indirect method to estimate survival from a reduction in movement for instance. Acoustic or radio telemetry and data-storage tags are novel and provide high-resolution data of discarded organisms in their natural environment. While highly informative, they are typically costly and therefore limited in observation coverage (Yergey *et al.*, 2012). These types of studies provide new insights on, for instance, sublethal effects or behavioural adaptations after being discarded (Neat *et al.*, 2009). The number of studies that has estimated long-term survival at fleet level are limited or maybe even non-existing (ICES, 2014c). Another approach to assessing long-term survival is the estimation of factors that compromise survival of organisms that are returned to their natural habitat, e.g. predation of seabirds on discarded organisms (PARTIM IV).

6.5 Ethical statement

Fish experiments were performed in accordance to the recommendations laid down in the Royal Decree (Belgisch Staatsblad. 29 Mei 2013. Koninklijk besluit betreffende de bescherming van proefdieren, bl. 42808). Animals were captured in the wild to be representative of commercial fishing practices. All efforts were made to minimize animal suffering, in particular in comparison to the stress experienced during the capture and discarding process of commercial fisheries.

6.6 Acknowledgments

I am indebted to the crew of RV 'Belgica' for logistics during sampling, to the ILVO-colleagues assisting during the trials and preliminary analysis (Eddy Buyvoets, Patrick Calebaut, Marieke Desender, Jolien Goossens, David Vuylsteke, Christian Vanden Berghe) and to the former commercial skipper (Erik Lagast) for a critical view. I also thank Michael Davis, Sebastian Uhlmann, Maarten Soetaert and Ruben Theunynck for their valuable suggestions as well as anyone at the ICES Workshop on Methods for Estimating Discard Survival for steering thoughts and recommendations.



*Can we
reduce
discarding?*

PARTIM III

Discards from human and stock perspective

7 Gear measures to reduce discarding

Doubt is not a pleasant condition, but certainty is absurd. – Voltaire

7.1 Change by not changing at all

A fishing gear constitutes the one and only physical interaction between humans and the marine ecosystem to fish for naturally produced biological food resources. The interactions are direct and all socio-economic and ecosystem effects of fishing are inherently and inevitably related to the deployment of the gear as a tool through which fishing interacts with the marine ecosystem. It may not be surprising that the ecosystem effects of fishing relate in one sense or another to the fishing gear:

- either through the physical properties of the gear which may affect the physical, (biogeo)chemical and bio-ecological attributes of the marine ecosystem and its components: the intensity of deployment of a 'generic' gear,
- either through the way a single fisherman uses it through time and space to exploit natural resources, i.e. the intensity of deployment of fishing gear tuned to the specific requirements of a fishermen in a specific situation,
- either through the way society allows the fishing fleet to exploit natural resources: frequency, time and location of deployment of 'generic' gears.

Fishing gears are the tools to deliver food from the marine ecosystem. This tool should be developed towards the 'best' way of interaction with the environment. Its physical properties are one of the most obvious elements to change in order to change the return from fishing on society and on the marine ecosystem (Suuronen *et al.*, 2012). However, it should be acknowledged that changing these properties primarily affects the intensity of interaction, whereas the effects on society and the marine ecosystem are also, and maybe more substantially, bound to its deployment (frequency, time and space). None of fishing gears is perfectly suited to deliver the requirements of society, as propagations of their direct effects are not fully understood; and both the environment and societal needs are dynamic. However, some fishing gears are better suited than others to achieve management objectives. Perfect gears may be defined as gears with minimal economic cost (mainly fuel), maximum social output (employment, safety, etc.), maximum marketable catch (highly selective) and a minimal impact on non-target species, marine habitats and the entire ecosystem.

7.1.1 *Why gear measures and do they work?*

Meeting management objectives requires change to the current fishing practices. Gears are a simple tool to bring about change while ‘not changing at all’. Acquiring fishermen to account for ecological objectives and invest in societal objectives rather than direct, short-term personal profits is a daunting task. Scientists have reasoned (‘top-down’) that minor modifications to a fishing gear may assist in meeting management objectives without the need for fishermen to drastically change their common practice (‘bringing change while not changing at all’). For decades, fishing gear technologists have attempted to predominantly change the physical properties of a fishing gear to reduce the discards without economic loss, i.e. without reduction in landings of the target and by-catch species. The general concept of fishing gear technology is to increase the size and species selectivity (Graham *et al.*, 2007; Kaiser *et al.*, 2007), and to reduce its physical interactions with ecosystem attributes which are not marketable (e.g. avoid seabed contact through off-bottom trawl doors) (Valdemarsen *et al.*, 2007; Valdemarsen & Suuronen, 2002). The reduction of discards of small, juvenile fish by gear selectivity measures is still highly topical, as Chapter 1 as well as others (Heath & Cook, 2015) highlight that the bulk (~60% in Heath & Cook, 2015) of the discards is related to undersized fish. The efficacy of gear measures to reduce discards varies considerably (Catchpole *et al.*, 2008a; Enever *et al.*, 2009; Favaro & Côté, 2013; Graham *et al.*, 2007; Nikolic *et al.*, 2015; Raveau *et al.*, 2012). Clear examples where the common fishing practices have not changed much, but where the implementation of and compliance with these measures have led to effectively reduced impacts are the introduction of the sieve net in shrimp beam trawling, the use of off-bottom longlines to reduce elasmobranch bycatch, the use of bird-scaring lines matched with weighted branch lines in longline fisheries (Catchpole *et al.*, 2008b; Favaro & Côté, 2013; Melvin *et al.*, 2013; Polet *et al.*, 2004; Willems *et al.*, 2013). Overall, gear measures have not reduced discards levels sufficiently to meet the management objectives, hence the reform of the Common Fisheries Policy.

7.1.2 *The development of gear measures*

Fishing gear technology is a particular branch of fisheries’ science which focuses particularly on the development of gear modifications. Fishing gear technologists may in this sense keep track of the continuous technological developments in the fishing fleets. Commercial fishermen generally focus on increases of the catch efficiency and may use technological developments to this end, whereas gear technologists combine technical developments for both the increase of catch efficiency as well as the reduction in ecological impacts. An example may be the replacement of the beam trawl shoes and beam by a ‘Sumwing’ to reduce drag during fishing. A reduction in drag implied that Sumwing trawls consume less fuel than conventional beam trawls to cover the same surface (‘swept area’)

(Beare & Machiels, 2012; Polet & Depestele, 2010). This development may reduce fishing impact on the seabed, because there is potentially less contact of the gear with the seabed. However, fishermen may use heavier chains, or fish at an elevated speed to cover more fishing grounds at the same fuel cost. The gear development may thus be perceived as a reduction in fuel consumption but its reduced seabed contact is questionable. Aside of the actual outcome of this example, it illustrates that both fishing gear technologists and fishermen are closely working together in the development of new gears or the modifications of existing gears, albeit with a potentially different objective. Fishing gear technologists and fishermen both have great expertise in assessing the fine-scale interactions between gear and the environment from close collaborations among them. Potential mitigation measures are developed on the basis of the exchange of knowledge. If science-fishermen partnerships are to work, then it may be expected within this field of expertise.

Because of their close collaborations with the industry, gear technologists may also provide a firm and up-to-date knowledge base on the catch efficiency of fishing vessels, preventing that fisheries assessment lack behind reality by ignorance of technological creep (Rihan *et al.*, 2011). Technological creep implies a mismatch between fishing effort descriptors and recent developments of the fishing efficiency of gears through for instance newly developed groundropes, sizes of the nets, type of catch stimuli in beam trawls, etc. (Eigaard *et al.*, 2014; Marchal *et al.*, 2006; Reid *et al.*, 2011; Whitmarsh, 1990). Developing fishing gears measures is thus not only of interest for its direct outcome.

7.1.3 Implementation of gear measures and the role of fishing gear technologists

Many aspects are of importance when developing new technologies, not in the least the uptake of the new developments by the fishing industry. Gear trials are generally focused on small-scale experimental trials which may create scope for reduced impact in particular environmental situations and gear riggings, but not in others. While this may hamper uptake, the loss of income is likely of even greater importance for the uptake of new developments. If gear measures 'work', the greatest benefits of the new technology may be achieved when embedded in the management system or when there are clear incentives to use and develop them (either legislative or voluntarily through increased profits) (Ainsworth *et al.*, 2012; Jennings & Revill, 2007). This particular element may have been lacking in many newly developed gear modifications, as implementation of gear modifications is a continuous topic of high debate among gear technologists (ICES, 2014d). A lack of implementation of gear measures, or when implemented, a lack of compliance and/or effectivity, is in many instances apparent. It is concluded that gear technology provides satisfying results when fishermen are incentivised and supportive to its application (Diamond & Beukers-Stewart, 2011).

7.1.4 Gear measures in Belgian beam trawl fisheries

Advances in the reduction of ecological impact of the Belgian fishing fleet between 2004 and 2014 were particularly resulting from (1) fleet reduction (decommissioning), and (2) gear modifications and/or replacements to reduce fuel consumption (stimulated by the rising fuel prices in 2008). The Belgian fleet has reduced from 125 fishing vessels in 2004 to 79 in 2014. Beam trawling was and is the most frequently used fishing gear, although other gears were also used: 25 otter trawlers and four 'other' gear types in 2004 (DLV, 2005). The number of otter trawlers dropped in 2010 to four while the number of gill netters increased to seven. The rising fuel prices were a stimulus towards the use of passive gears, but the gill netters did not succeed in making profit and their number dropped back to two in 2014. The number of otter trawlers increased again to 17 in 2014. The shifts in fishing gears are likely the most apparent changes in ecological impact from a gear perspective. Aside gear replacements, modifications were also introduced in the beam trawl fisheries. Gear modifications focused on reducing fuel consumption by reducing seabed interactions. They included the use of lighter beam trawls, Sumwings, outrigger trawls, etc. (Platteau *et al.*, 2014; Roegiers *et al.*, 2013; Vanderperren, 2008). Gear modifications to increase selectivity were during this period not introduced, except for a panel of larger mesh sizes in the top panel (DLV, 2013). Its efficacy has not been evaluated at fleet level, although a reduction in discards of whiting and haddock may be expected (Fonteyne, 1997).

Discard mitigation research has, in contrast, also invested in gear modifications to increase species and size selectivity (Depestele *et al.*, 2008b; Depestele *et al.*, 2008c; Fonteyne & Mrabet, 1992; Fonteyne, 1997; Keegan, 2002; Polet, 2007; Van Marlen *et al.*, 2010). The developed gear modifications, however, are hardly used by the fishing industry. While fishing gear research has advanced, the major gap seems to be the implementation of the gear modifications in the fishing fleet. Selectivity is nevertheless opted as one of the primary mitigation measures for the Belgian fleet with the introduction of landing obligation (DLV, 2013; SALV, 2014). The landing obligation is expected to stimulate the use of gear modifications in the Belgian beam trawl fishery. The most recent efforts undertaken to implement the use of gear modifications in the Belgian fishing fleet are proposed in a collaborative effort of fisheries managers, the fisheries research institute, the producer's organisation of the fishing industry and a non-governmental organisation (De Snijder *et al.*, 2014), and potentially a labelling system to compare the relative sustainability for each participating fishing vessel (Kinds & Sys, 2015). Results are expected in the coming years.

Gear research efforts on reducing discards are evaluated by their capacity to do so, as well as their capacity to maintain the catches of marketable sole, the most valuable fish species in Belgian

fisheries. Gear modifications can be grouped into three types of gear measures: (1) net modifications to beam trawls (changing net selectivity), (2) modifications of the catch stimuli (changing catch selectivity) and (3) replacing beam trawls by other gears (changing exploitation patterns).

1. Modifying the net of beam trawls

The net of a beam trawl generally consists of a belly, a top panel, two side panels and the codend. Gear selectivity of the net may be realised in a variety of ways, and has already been tested for different compartments of the net. Combinations of these modifications potentially lead to an additive effect, although the behaviour of the net as a whole and its water flow may lead to unexpected outcomes and should therefore be tested prior to taken for granted. Different mesh sizes, colours, shapes and materials may be used for each of the compartments, as well as changes to the overall shape of the gear, and/or the use of sound, odour, electrical, light and physical stimuli in several parts of the trawl net (ICES, 2005). The most obvious difference is the contrast between a V-net and a R-net. Both nets are used for beam trawling, although with different catch stimuli: tickler chains and chain mats respectively. The latest development relates to a square opening of the net, which is typically used in pulse (beam) trawling. A brief summary of modifications to the different net compartments will be listed with a reference to the full information source.

a) Selectivity of the codend

A conventional beam trawl codend for sole-directed fisheries consists of diamond meshes of 80 mm mesh size. The selectivity of the codend is specifically designed to retain sole of the minimum landing size (MLS: 24 cm), which results in a reduced size selectivity for other species, notably plaice which has a MLS of 27 cm (Figure 7.1). When the diamond mesh shapes are replaced by square meshes or T90-meshes (meshes which are turned 90°), the shape of the selectivity curves becomes steeper for flatfish and flattens for roundfish (Fonteyne & Mrabet, 1992; Van Craeynest *et al.*, 2013). These opposite effects are the result of the opposing shape of the meshes when tension builds up in the codend. Square or T90-meshes become rounded, but diamond meshes get an elongated diamond shape when the catch is piling up in the codend (Figure 7.2). The selectivity of the T90-codend reduces the loss of marketable sole, but also increases the number of undersized soles. An introduction of a square mesh or T90-codend may therefore require an increase in mesh size. The stability of the mesh shapes of the square mesh and T90-codends may require further investigations, as they tend to deform after multiple hauls (Depestele *et al.*, 2008b).

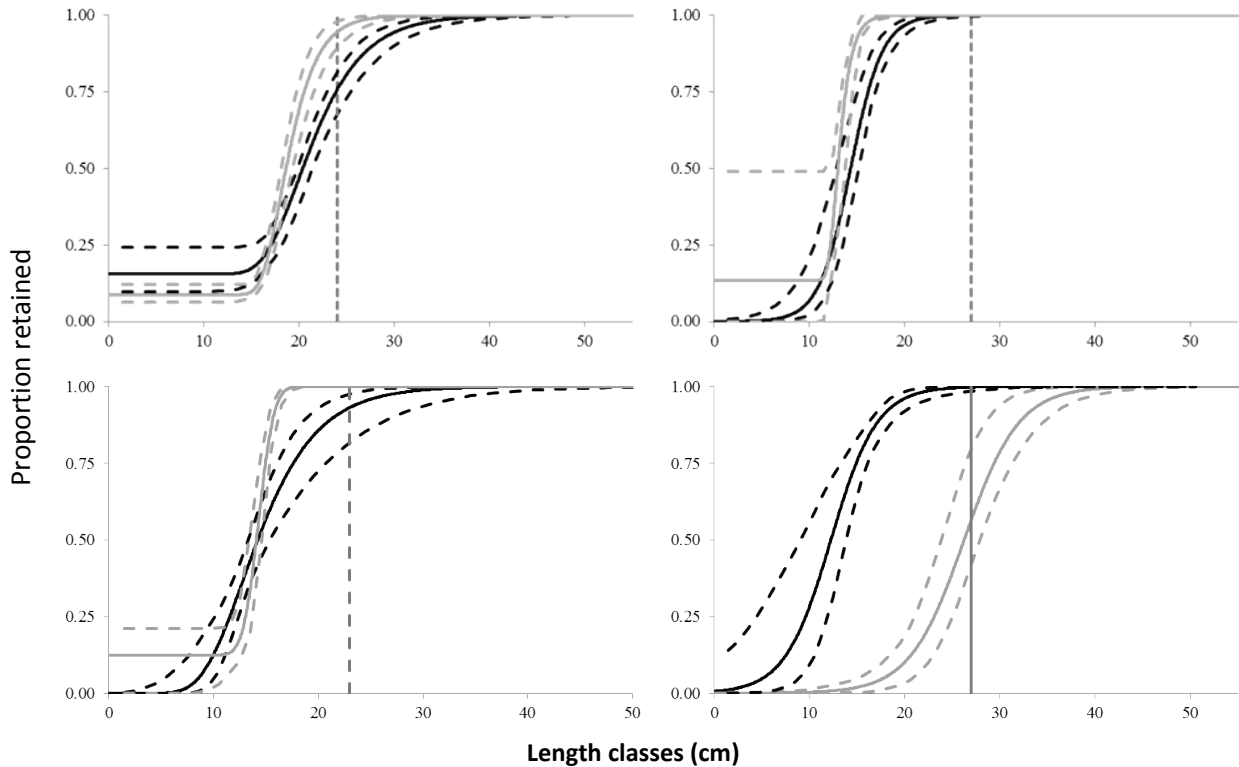


Figure 7.1 Size selectivity curves for sole (upper left, MLS: 24cm), plaice (upper right, MLS: 27cm), dab (lower left, MLS: 23cm) and whiting (lower right, MLS: 27cm) in a 80 mm beam trawl codend with diamond-shaped mesh (black) or T90-shaped mesh sizes (grey). Dashed lines indicate the 95% confidence intervals, while the vertical line indicates the MLS (after Bayse *et al.*, 2015).

b) Changes to the belly panel

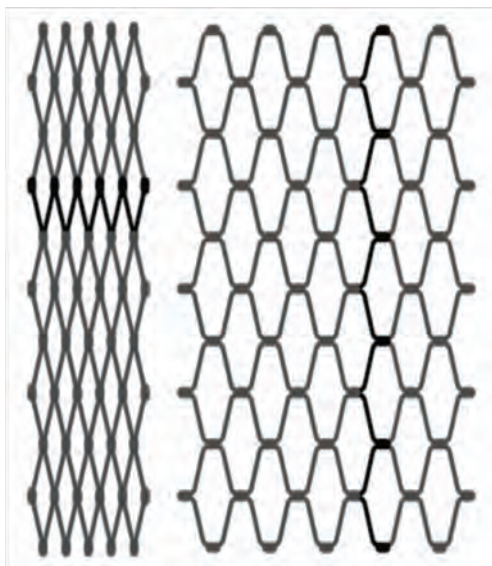


Figure 7.2 Mesh shape of diamond (left) and T90 (right) meshes. T90: Turned 90 degrees.

Discards of benthic invertebrates can be reduced by modifying the belly panel of a beam trawl where heavier and larger benthic invertebrates may drop through (Depestele *et al.*, 2008b; Fonteyne & Polet, 2002; Reville & Jennings, 2005). A zone without netting behind the groundrope reduces the commercial catch of sole significantly, but a square mesh window in front of the codend (benthos release panel: BRP) only affected the catch of sole to a limited extent, while 83% of the benthic invertebrates were lost. The loss of sole may be prevented using the BRP with electric pulses (Soetaert *et al.*, 2014a).

c) Changes to the top panel

Roundfish species such as whiting and haddock tend to move towards the upper part of the trawl when being caught, while flatfish exhibit a diving behaviour. The use of large meshes (>100cm) in the top panel of tickler chains beam trawls indicated that cod and whiting catch could be reduced with a minimal loss of sole and plaice. Length-frequency indicated (visually) that the reduction in roundfish catches was indifferent from lengths (Van Marlen, 2003).

The large mesh panels used in the Belgian beam trawl fishery have a mesh size of 36cm, indicating that the loss of roundfish will be less, although the meshes are likely sufficiently large to let roundfish escape through the top panel, especially since beam trawling with chain mats occurs at a lower speed. The expected increased selectivity is thus limited. Square mesh panels and a cut-back cover were tested in chain mat beam trawls and significantly reduced discards of whiting and haddock between 5% and 48%, but were inconclusive for cod (Fonteyne, 1997). The catch reduction was more pronounced for larger vessels, indicating the time that roundfish needs to escape.

d) Introduction of additional compartments using separator panels

Several types of separator panels exist: inclined, inserted to the belly of the beam trawl or between the riggings of the upper and lower panel. The primary objective of these panels is to utilize the behaviour of the fish such as upward or downward escape movements to create different compartments in the trawl. If species can be separated from one another, they may end up in different codends with different mesh sizes. Investigations of separator panels in beam trawls are predominantly directed towards the separation of sole from all other fish species, especially flatfish such as plaice. When sole may be caught in the lower compartment, as sole is expected to dive and dig in the seabed after being disturbed, and when plaice and dab end up in the upper codend, different mesh sizes may be used to have the appropriate size selectivity for sole in the lower codend (80 mm) and for plaice in the upper codend (e.g. 120 mm, Polet *et al.*, 2010). Several tests have been conducted (Goossens, 2014; Males, 2012; Van Craeynest *et al.*, 2013), but none of them has resulted in a 'sufficient' catch of sole in the lower compartment. Several lengths of the panel were tested, several mesh sizes and shapes, but a complete separation of sole could up to now not be obtained. The 'best' separation retained 15% of marketable sole in the upper codend. The use of a separator panel for flatfish is subject to further research, but may require input from knowledgeable fishermen to reach an 'acceptable' level of separation (Visserijnieuws, 2013).

A limited number of trials indicated that the reduction of discards by net modifications has potential for certain species. The discards of some roundfish species (whiting and haddock) may be reduced through different mesh shapes in the codend (small fish) and escape openings through the top panel (all sizes). How cod discards may be reduced is less clear. Gear modifications to reduce the discards of benthic invertebrates show potential as well, although it has long been accepted that beam trawling induces the highest benthic mortality rates in the tow path rather than in the catch, rendering the expected merit of this modification limited (de Groot & Lindeboom, 1994; Lindeboom & de Groot, 1998; Revill & Jennings, 2005). While recent developments (pulse trawling) reduced the physical interactions between some gears (e.g. pulse trawl) and the seabed (Depestele *et al.*, 2015) and potentially the catch efficiency for benthic invertebrates, a revision of the balance between mortality in the tow path and in the catch is required to evaluate the potential reduction in mortality by BRPs, i.e. catch efficiency should be evaluated. The most obvious bycatch in sole-targeting beam trawl fisheries are other flatfish species. Flatfish have similar mechanisms to escape predation or other threats. An effective net modification has up to this date not been developed, although there is potential for a reduction in discards if a 'limited' loss of sole is accepted, or if only a limited discard reduction of dab and plaice is deemed sufficient (Van Marlen *et al.*, 2013).

Net modifications in beam trawling have been investigated in detail during several decades of Belgian as well as Dutch research projects, and have shown potential to reduce discards. Successful implementation of these net modifications, however, depends to a great extent of fishermen's involvement and active participation. There are two net modifications which have been implemented in the Belgian fishing fleet: (1) larger meshes in the top panel (diamond mesh: 36 cm) to reduce roundfish discards, and (2) the use of a large mesh extension in the trawl to reduce the catch of undersized sole (Bayse & Polet, 2015). The latter gear modification was introduced in April 2015 to accommodate for a proposed 60% reduction in sole quorum in ICES Division VIIId. The Belgian fishing industry proposed to use a trawl extension of 120 mm mesh size to reduce the discards of juvenile sole so that they could prevent the sole quorum to be reduced to 60%. The quorum was reduced by 28% instead.

2. Modifying the catch stimulus of beam trawls: pulse trawling

Discard mitigation measures do not only result from changes in selectivity of the net of a beam trawl. The catch efficiency may be a more effective way to reduce discards; and especially because it does not include the uncertainty of mortality or reduced fitness from individuals that escaped through the meshes of the net. Several mechanisms were tested to catch sole with a reduced physical disturbance of the tickler chains. Primary investigations focused on the number of chains in tickler

chain beam trawls: the catch efficiency increases in sandy habitat with the number of chains for burrowing species or species that are in close association with the seabed, but does not change significantly in muddy habitats (Creutzberg *et al.*, 1987). A different arrangements of the chains were tested (longitudinal to towing direction), but not found effective in catches sole and reducing discards of benthic invertebrates (Keegan, 2002). Catching sole by mechanically stimulating them by changing the water flow was tested using two types of water jets (Keegan, 2002) or metal cups (unpublished trials on-board RV 'Belgica'; van Duren & De Kleermaeker, 2011; Van Marlen *et al.*, 2011b), but did not result in an increased catch efficiency of sole relative to other organisms.

The replacement of tickler chains (and/or chain mats) by electrodes to fish with electric instead of mechanical stimuli, however, was found effective for catching sole (Rasenberg *et al.*, 2013; Van Marlen *et al.*, 2014). The catch efficiency of sole is higher in a pulse (beam) trawl than a conventional beam trawl with chains, while the rigid backbone of plaice prevents it from being caught as effectively as sole in pulse trawls. Both, sole and plaice exhibit a cramp reaction when they are in the electric field. The resulting U-form from contracting their powerful dorsal muscles is more pronounced for sole than plaice, which indicates the higher relative catch efficiency for sole than for plaice in pulse trawling than in conventional beam trawling. This mechanism is suited for separating sole and plaice in beam trawl catches to a certain extent. Pulse trawling holds the potential to reduce the discards of plaice while still catching sole (Rasenberg *et al.*, 2013). The reductions in fuel consumption by pulse trawlers and the increased catch efficiency for sole renders this gear development very profitable (Soetaert *et al.*, 2013), which is clearly demonstrated in the competitive advantage of Dutch fishermen to the disadvantage of Belgians (Sys *et al.*, 2015).

Using electricity to catch the target species is highly effective catch stimulus which reduces the discards, but also changes the overall ecological effects of the fishing gear. The direct changes are related to the catch stimulus, but other effects include changes in fishing behaviour (towing speed, frequency of repeated trawling, etc.) and exploitation pattern (fishing effort, location and time of fishing). The pulse (beam) trawl may be considered as a different metier (depending on its definition) than conventional beam trawling (which is being differentiated by mesh sizes but not by catch stimuli such as tickler chains and chain mats). A brief and non-exhaustive overview of the direct changes in ecological effects includes (1) changes in catch composition, (2) effects of electricity (3) habitat effects, and (4) changes in exploitation pattern. Note that all statements below are based on a limited number of records or peer-reviewed papers from a small sample (e.g. limited number of experimental trips and vessels, high-resolution but local investigations, etc.), and should therefore not be taken for granted at fleet level, but interpreted carefully.

a) *Changes in catch composition*

The total catch volume of pulse trawling is reduced in comparison to tickler chain beam trawling. However, while the total catch weight of marketable fish decreases in Van Marlen *et al.* (2011a, 2014), Rasenberg *et al.* (2013) noted that the plaice landings in pulse trawling are lower than conventional beam trawling. The sole landings however increased in pulse beam trawling. The absolute landings of plaice however are higher than those of sole. The trends are more pronounced for marketable fish than for smaller individuals. The lower catch efficiency of small sole and small plaice increases the potential of reduced discards of small organisms of both species. Overall, the pulse trawl discards were around one third of the total weight of the discards of a conventional beam trawl, and plaice discards were three times less per kg landed sole comparing pulse to tickler chain beam trawlers (Van Marlen *et al.*, 2014). Approximately half (44%) of the catch weight of pulse trawling is landed (excluding debris), implying that >50% of the catch is discarded (fish: 23%; benthic invertebrates: 33% of the catch). The reduction in number of crustaceans and echinoderms discarded in pulse trawling indicates the reduction in invertebrate catch clearly (Rasenberg *et al.*, 2013; Teal *et al.*, 2014). While a detailed description of the catch composition is not available, the trend is clearly indicating a reduction in discards of both fish species (small target species, all species pooled) and invertebrates, while maintaining and likely increasing the catches of marketable sole. It is paramount to verify whether the overall reduction in catch composition is based on all caught species or exclusively on species that are related to the catch stimulus. Are the catches of cod and whiting for instance reduced by the replacement of electrodes with tickler chains or does it not affect its catch?

The suite of marketable species has not changed considerably by replacing beam trawling with pulse trawls. However, the relative catch efficiency was altered substantially to the advantage of the most valuable species, sole. This development increases the potential to reduce discard composition in beam trawling, especially when combined with net selectivity. While the gear shows potential for discard reduction, any changes to other pressures on the ecosystems should be investigated as well, to prevent that reducing one particular pressure does not go at the expense of another pressure.

b) *Effects of electricity on marine organisms*

The knowledge of adverse effects of electricity on marine organisms is scarce, although an increasing number of studies is being conducted in the lab as well as in the field to investigate possible adverse effects (Rasenberg & Rijnsdorp, 2014; Soetaert *et al.*, 2013). The possible adverse effects of electricity are numerous, such as effects on physics, physiology

and behaviour of marine organisms (e.g. which organisms are sensitive? sensitivity of early life stages?) and effects on geo-chemistry (electrolysis and release of nutrients or contaminants). The effect of a single exposure of cod, sole, dogfish, shrimp and *Nereis* sp. to electric pulses did not result in mortalities in the lab, nor was feeding behaviour of the fish species affected (Desender *et al.*, 2015; Soetaert *et al.*, 2014b). A causal relationship between the increased occurrence of ulcers in dab and the increasing effort of pulse fisheries could not be demonstrated (Devriese *et al.*, 2015). Studies are on-going, but the current indications point out that the effects seemed to be small.

c) *Habitat effects*

The replacement of tickler chains by electrodes may cause changes in the effects of beam trawling on the physical, biological and geochemical aspects of benthic habitats. Mortality in the tow path was investigated in Teal *et al.* (2014) by investigating the densities of benthic invertebrates caught in a benthic sledge before and after trawling, while accounting for possible density changes due to other, non-fishery related causes (BACI-study, Before-After Control-Impact study). The densities of individual species were highly variable and did not show a consistent pattern, i.e. the biomass did decrease after trawling, but this decrease was also noted in the reference area, which did not experience trawling. Most species (e.g. *Ensis* sp., *Abra alba*) had life history traits which indicated that they were resistant to trawling, and may be a likely explanation for the absence of tow path mortality. The biological effects of conventional beam trawling were clearly demonstrated, but the expected effect of pulse trawling on mortality of benthic invertebrates in the tow path remains unclear.

The physical effects of beam trawling and pulse trawling have been investigated in a single experiment in the Voordelta (Depestele *et al.*, 2015). The physical impact was investigated of both a conventional 4 m tickler-chain beam trawl and a “Delmeco” electric pulse beam trawl. The changes in seabed bathymetry were measured following the passage of these gears using a Kongsberg EM2040 multi-beam echosounder and use a LISST 100X particle size analyser to measure the concentration and particle size distribution of the sediment mobilized into the water column. The penetration of the gears into the seabed was estimated using numerical models for the mechanical interaction between gears and seabed. The results indicate that the seabed bathymetry changes between ~1 and 2 cm and that it is further increased by higher trawling frequencies. Furthermore, results suggest that the alteration following the passage of the conventional trawl is greater than that following the pulse trawl passage. There was no difference in the quantity of sediment mobilized in the wake of these two gears; however, the numerical model introduced in this study predicted

that the tickler-chain trawl penetrates the seabed more deeply than the pulse gear. Hence, greater alteration to the seabed bathymetry by the tickler-chain beam trawling is likely to be a result of its greater penetration. Further investigations are required to extrapolate the findings to fleet level, as pulse trawls as well as physical effects vary among fleet segments and sediment types.

d) *Changes in exploitation pattern*

Gear effects vary spatially and temporally. Comparing of the effects of pulse and beam trawling requires that both fisheries are conducted during ‘normal’ (commercial, *in situ*) practice. There is very little information on ‘normal’ practice of pulse trawling, as this fishery is being developed over the last couple of years. One important aspect of this practice is its spatial distribution. The distribution of the Dutch fishing fleet has remarkably changed since the introduction of pulse trawling (Rasenberg & Rijnsdorp, 2014). The fishing grounds of the pulse trawlers is located in the southwest of the southern North Sea, while the Dutch ‘tickler chain’ beam trawlers primarily fish in the eastern part of the southern North Sea (see Figure 7.3 for more indications of shifted distribution). The competition between Belgian and Dutch fisheries has changed increasingly, as Belgian fishermen were broadly located in the southwestern part of the southern North Sea (Sys *et al.*, 2015). Changes in exploitation pattern may result in changes in catch composition as well as changes in habitat effects, other than those measured directly in experimental conditions.

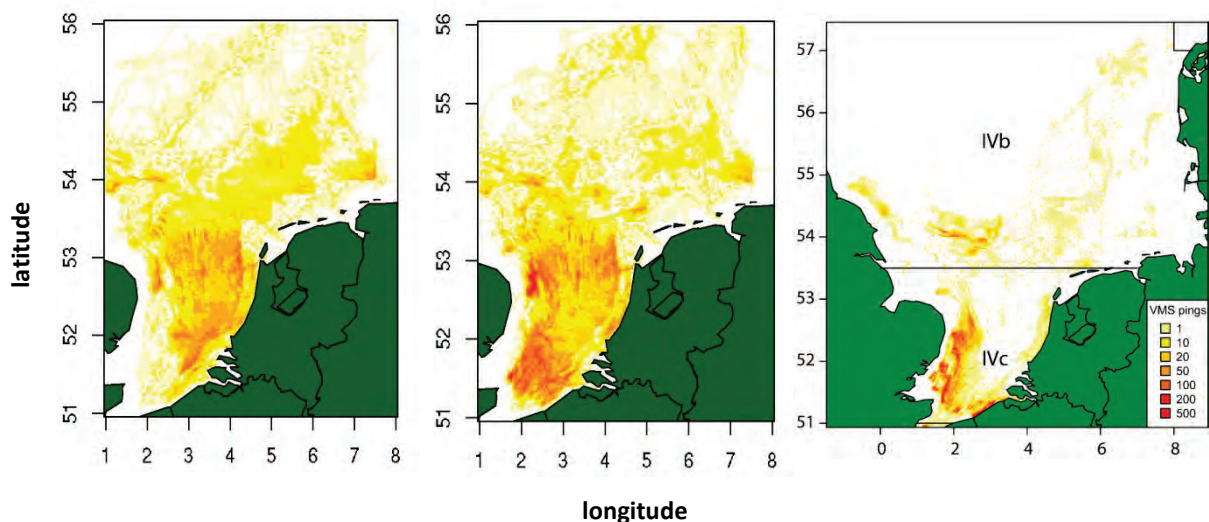


Figure 7.3 Spatial distribution of Dutch ‘tickler chain’ beam trawlers (left), Dutch pulse trawlers (middle) and Belgian beam trawlers (right). The maps are based on VMS-pings, and are indicative of the spatial distribution of beam trawling in the North Sea. After Rasenberg & Rijnsdorp (2014) and Vanellander *et al.* (2015).

3. Replacing the beam trawls by 'other' fishing gears to catch sole

To the far end of the spectrum of gear-based mitigation measures is the replacement of one gear by another. Catching sole may also be profitable with gill nets *sensu lato* (*s.l.*: gill and trammel netters), as is clear from French, Dutch and Danish fisheries. Trammel netting is different from gill netting, as it consists of three walls (instead of one) of multifilament or monofilament netting. The inner wall has smaller meshes which hang loosely in between two walls of larger mesh sizes. Gill nets catch fish by wedging, gilling or entanglement (catching organisms by their spines or other protrusions), while trammel nets also catch organisms by pocketing them. Pocketing implies that fish or invertebrates are caught in a pocket formed by the inner wall being pushed through one of the outer walls. Both, gill nets and trammel nets are effective in catching sole, albeit the catch efficiency of trammel nets is higher than for gill nets (Marchal *et al.*, 2007; Marchal, 2008). A small number of gill netters has been profitable in Belgian fisheries during the last decade, focusing on a range of target species: sole, cod and sea bass (Depestele *et al.*, 2006b). Trammel netting was proposed as an alternative to beam trawling from sole during the years following high fuel prices (Depestele *et al.*, 2008a; Depestele *et al.*, 2006a; Stouten *et al.*, 2011; Van Craeynest *et al.*, 2013), and a number of trammel netters were introduced in the Belgian fleet, although unsuccessful.

Trammel nets are considered less size-selective than gill nets. The length-frequency distribution (LFD) of trammel nets is skewed to the right, i.e. catching larger individuals (Erzini *et al.*, 2006). The species selectivity of trammel nets is less than those of gill nets (Stergiou *et al.*, 2002; Stergiou *et al.*, 2006; Suuronen *et al.*, 2012).

Overall, gill and trammel netters are considered more size selective than bottom trawlers. The (theoretical) size-selectivity curves of trammel nets indicate that a smaller length range is caught for sole, plaice, cod and whiting, than is the case in beam trawling (Figure 7.4) (Holst *et al.*, 2002; Madsen, 2007). The accuracy of size-selectivity curves of gill nets are under on-going debate, although some sort of bell-shaped function with a tail may be considered realistic to the actual selectivity (Brenden & Zhao, 2012; Smith & Taylor, 2014). While an S-shaped function is widely accepted in bottom trawl fisheries, the expectation of catching all individuals above a certain size is unrealistic (Rochet *et al.*, 2011). Surveys by underwater cameras have indicated that the selectivity ogives of bottom trawls may also be bell-shaped (Wells *et al.*, 2008) and the increased catch efficiency of pulse trawls in comparison to conventional beam trawls also indicates that the catch efficiency of bottom trawls cannot be 100%. Trawl selectivity studies are generally restricted to the codend, as fish escape is generally expected through the codend rather than the belly (Madsen, 2007). The contrast may be expected, however, from the gear modifications listed above (top panel

selectivity, pulse trawling). The differences between selection ogives of beam trawls and gill nets should be interpreted carefully, although they do indicate expected differences in size selectivity from experimental gear trials (Figure 7.4).

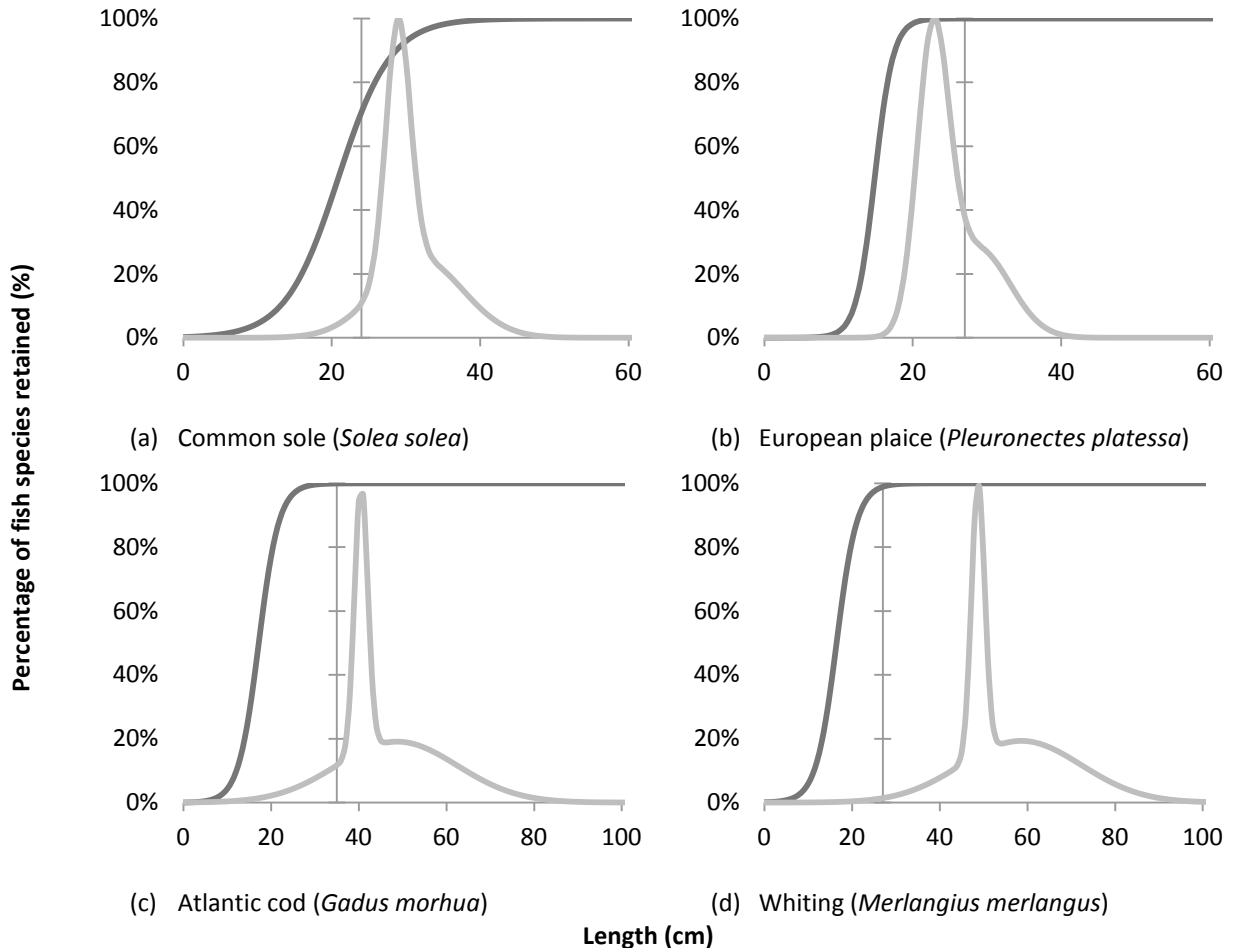


Figure 7.4 Selectivity ogives for sole, plaice, cod and whiting for two sole-targeting fishing gears: beam trawl with 80 mm diamond mesh in the codend (dark grey) and gill nets with 90 mm mesh sizes (light grey). The Y-axis intersects the X-axis at MLS: 24 cm (common sole), 27 cm (European plaice and whiting), 35 cm (MLS for Atlantic cod). MLS: Minimum Landing Size.

Empirical data illustrate that the mean length of gill net catches s.l. are higher than those of bottom trawlers, which implies that bottom trawlers catch more smaller fish (Erzini *et al.*, 2002; Fauconnet *et al.*, 2015). In contrast, the length ranges of the bottom trawl catches were smaller than those in gill nets (Depestele *et al.*, 2012: 153-154; Fauconnet *et al.*, 2015). Discard rates in the Belgian trammel net fisheries were estimated from a limited number of observer trips (see Depestele *et al.* (2012) for details). The observed discard rate (discard weight / catch weight) were 0.56 (SD: 0.45) for dab, 0.57 (SD: 0.48) for flounder, 0.70 (SD: 0.35) for plaice and 0.02 (SD 0.03) for sole. The discard rate was 0.22 for all fish species combined. The high variability in discarding indicates differences in fishermen's behaviour and spatio-temporal characteristics.

The discards in sole-targeting trammel net fisheries are also high. However, a direct comparison between gears was not possible due to confounding effects of fishing grounds (Fauconnet *et al.*, 2015). Belgian trammel net fisheries were conducted in locations which were inaccessible to beam trawlers or when trammel net fishing took place in the gullies between the sandbanks in the Belgian Part of the North Sea (BPNS), trammel net fishermen were protecting their nets from being trawled away. The expected potential reduction in discards of plaice and other flatfish species by replacing beam trawls with trammel nets, however, did not seem to be the case in the BPNS.

The discards of non-commercial species were compared between beam trawls and trammel nets in the southern North Sea after being standardized to the catch weight of landed sole (discards (kg)/landings of sole (kg)) (Depestele *et al.*, 2012; Hostens *et al.*, 2014). Overall, the amount of non-commercial discards as well as the number of species was higher in beam trawl fisheries than in trammel net fisheries. The variability in discard composition was mainly explained by the gear, besides longitudinal position and depth, although most of the variation (65%) remained unexplained. Fauconnet *et al.* (2015) confirmed that the variability in species composition of the discards is difficult to explain by gear and location. The comparison of the discard composition between beam trawl and trammel net fisheries indicated a lower discard rate for benthic invertebrates, as well as a reduced species composition in trammel net fisheries. The reduction in discards of benthic invertebrates by replacing beam trawls with trammel nets, however, cannot be unambiguously concluded. First, the gear does not explain much of the variability, and second, there are potential confounding effects from the differences in sample sizes (Fauconnet *et al.*, 2015).

In conclusion, the reduction in discards by replacing beam trawls with trammel nets does not seem to hold much potential for commercial fish species. The reduction in discards of non-commercial species may be achieved by shifting gears as experimental trials suggest. The replacement of one gear by another to reduce discards, however, should be balanced against all other potential side-effects of the gear shift. Trammel netting may, for instance, result in by-catch of marine mammals (Bjorge *et al.*, 2013; Larsen & Eigaard, 2014). A framework to compare all potential pressures for both gears may assist in balancing the outcome of this type of gear shift (Depestele *et al.*, 2014b).

7.2 Gear is but one of the possible management measures

The causes of discarding are numerous, making it difficult to predict or model them (Chapter 4). The notion of discarding originates from the concept of the spawn-at-least-once policy, which may be seen as the fundamental basis from which all issues related to discarding begin (Chapter 1.4.1). A move towards another exploitation concept such as balanced harvesting may in the far future lead to a paradigm shift on this matter. The current growing focus, however, requires that solutions are found to avoid or reduce discards.

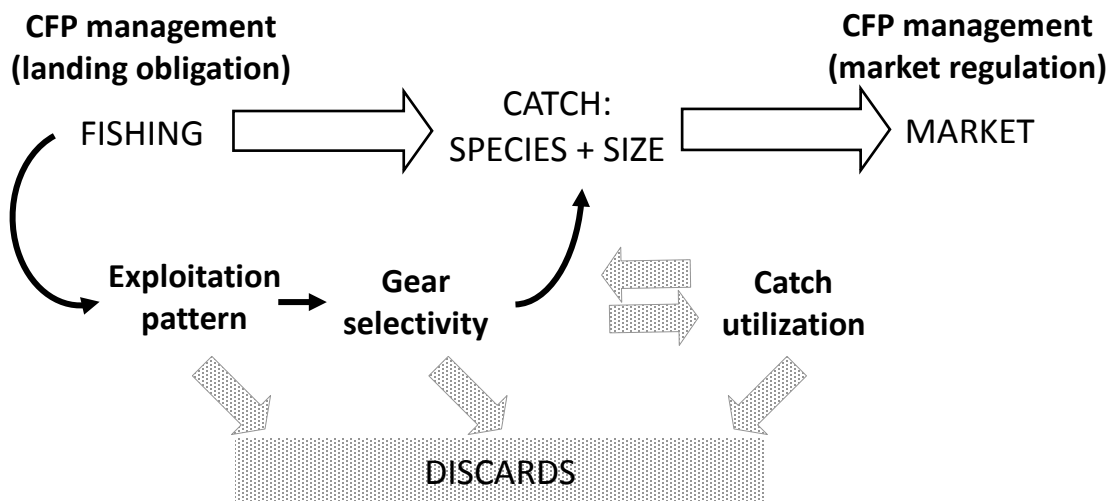


Figure 7.5. Scheme of the fishing process and tools to reduce discards (bold). Fisheries management sets the context for discarding through the landing obligation (EU, 2013a) and the market regulation (EU, 2013c). Fishermen may change discard levels mainly by their exploitation pattern and the selectivity of their gear. Discard levels are also influenced by the utilization of the catch by fishermen, which is strongly driven by the market. CFP: Common Fisheries Policy.

The tools to reduce discards are based (1) in the management system which sets the context, (2) in the way of exploitation of the natural resources and (3) in the utilization of the catches of fishermen (Figure 7.5). Discard reduction is predominantly directed towards changes in the way of exploitation, which is a combination of the fisheries' exploitation pattern and gear selectivity.

Other possible management measures than gear measures to reduce discarding are thus related to the exploitation pattern of fisheries. While gear measures are suggested to eliminate the catch of unwanted species that are being encountered, spatial and temporal patterns of fishing activities determine the encounter of unwanted species in the first place. High-resolution maps of the distribution of fish landings can be produced when VMS-records are coupled to logbook data. They may be coupled to distribution maps of discard data as well to create high-resolution maps of catch composition of different fisheries (Gerritsen *et al.*, 2012). Spatial and temporal distribution maps may also be produced for unwanted, vulnerable species (Cosandey-Godin *et al.*, 2014).

The spatial and temporal distribution of fish species and/or species of small sizes (spawning areas) may be used to prevent discarding by permanently or seasonally closing fishing grounds (Marchal *et al.*, 2002; Pastoors *et al.*, 2000; Van Keeken *et al.*, 2007). Closures are easy to implement and enforceable, but determining its size or timing is difficult, and the spatial heterogeneity and dynamic nature of species distribution complicates the designation of closures for discard mitigation even more. The failure of the 'Plaice box' may be a clear example of the limited return from stationary closures (Little *et al.*, 2014; O'Keefe *et al.*, 2014).

However, avoiding the encounter of unwanted species or species sizes holds great potential. Real-time management systems are being developed to inform fishermen through fleet communication systems where areas of low discards are located (Dunn *et al.*, 2011; Dunn *et al.*, 2013; Needle, 2015; Vilela & Bellido, 2015). The development of real-time spatial management system to reduce discards may be complemented with the management of other ecological effects of fishing, such as habitat impacts (Kraak *et al.*, 2014; Kraak *et al.*, 2012). The development of real-time, spatial management systems bears the powerful capacity to account for the highly dynamic nature of ocean environments, and is less prone to uncertainties such as escape mortality. Real-time, spatial management is based on the cooperation with fishermen to transfer information, and increases the responsibility of fishermen (results-based management, e.g. Nielsen *et al.*, 2015).

While high-resolution changes in the exploitation pattern and gear selectivity measures are related to changes in the fishing process, other measures may change the context of fishing and thereby also affect discard levels. Current discarding practices are partially driven by quota and MLS restrictions (Chapter 4), which may be reduced by changing the management system, e.g. by prohibiting highgrading (EU, 2013b). The current move of the European CFP towards an obligation to land all catches of a restricted list of species can be seen in this perspective.

Whether the elimination of fishery discards will also result in reduced fishing mortality, and particularly mortality of small fish, will depend on the way that catch quota will be set and enforced. The level of quota increase and enforcement will determine the incentive of fishermen to match their catch composition with the target catch levels (Condie *et al.*, 2013). A limited increase may stimulate fishermen to target larger species as prices of small fish will be yield less profit. The level of increase of landings quota is to create an economic incentive for fishermen to transfer to the catch quota system and to reduce the catches of small or unwanted fish (Rochet *et al.*, 2014b). The effectiveness of this management system as a stand-alone measure has been questioned (Condie *et al.*, 2013; 2014), although the actual result can only be tested from a real-time situation, even though reality may not be the appropriate place to conduct large-scale experiments.

While the landing obligation envisages the elimination of discards by landing them, it requires that the landed discards do not create new markets for fish meal, which may otherwise create a complementary income for fishermen and, in turn, generate a perverse effect on fishing mortality (Sardà *et al.*, 2015). The landing obligation was therefor complemented with a market regulation (EU, 2013c), which specifies that minimum marketing sizes should correspond to minimum conservation reference sizes, in accordance with Article 15(10) of the landing obligation (EU, 2013a).

While the new CFP is directed towards the elimination of discards to rebuild fish population, Heath & Cook (2015) mentioned that a further reduction of fishing effort may be another plausible management measure to reduce discarding. The study by Heath & Cook (2015) concluded that size is the main driver for discarding of the main target species. Reducing over-fishing and restoring fish populations to a state in which they contain a higher proportion of large fish was therefor proposed as the most effective remedy for discarding. The findings of Heath & Cook (2015) may be put into perspective by Catchpole *et al.* (2013) among others (Eliassen *et al.*, 2015; Kell & Bromley, 2004; Poos *et al.*, 2010; Chapter 4) by stating that other drivers than size contribute equally to discarding. Catchpole *et al.* (2013) showed that all four identified causes of discarding were substantial contributors: (1) fish below MLS; (2) fish for which there is no market and that do not have a MLS; (3) fish for which there are inconsistencies in market and sorting practices and (4) fish with quota restrictions. Their relative contribution varies by fisheries, as defined by country, areas, gear and species. The potential to reduce discarding by a further reduction in fishing effort may primarily relate to the reduction of discarded proportions rather than absolute levels of discarding.

*Towards the
quantification
of the fate
of discards*

PARTIM IV

Discards from ecosystem perspective

8 Partitioning discards between birds and scavengers in the sea

Collaborative study within the FP 7 Project 'Benthis' submitted to the Canadian Journal of Fisheries and Aquatic Sciences.

Depestele, J., Rochet, M.J., Dorémus, G., Laffargue, P. and Stienen, E. (submitted). Are discards a substantial food source for marine scavengers? Submitted to the Canadian Journal of Fisheries and Aquatic Sciences.

8.1 Abstract

Fisheries' discards subsidize seabird populations, but estimates of their contribution to the food requirements of marine scavengers in the water column or on the seabed vary from negligible to substantial. Variation in discard amounts and composition through space and time is one plausible explanation for these differences, though rarely accounted for. A framework was developed to include the spatial and temporal variation in seabird distribution, seabird attraction to fishing vessels and discard distribution in order to estimate discard consumption by seabirds over space and time. The framework was applied to the Bay of Biscay and showed high variation in discard consumption by seabirds across seabird foraging guilds, discard types, semesters and locations. The Bay of Biscay case-study showed that seabirds remove around one quarter of all discards; the remaining discards have limited potential to subsidize scavenging benthic communities on a large scale. Sinking discards may provide a substantial contribution to the food items of certain scavengers on a local scale though.

Keywords: discard consumption, discard mortality, discard partitioning, food subsidies, scavengers, seabirds

8.2 Introduction

Fisheries discards are a major food source for seabirds and significantly affect seabird ecology (Bicknell *et al.*, 2013). Marine mammals as well as scavengers occupying lower trophic positions have also been observed scavenging from discards floating on the sea surface or in the water column (Hill and Wassenberg, 2000), but data are scarce and the significance of discards as a food source is unclear at population level. Once discards have reached the seafloor, they may be consumed by demersal fish and benthic invertebrates. Discards in the North Sea were estimated to deliver 1-3% of the annual secondary production of the macrobenthic community, limiting its potential to cause direct population effects (Groenewold and Fonds, 2000). Another study in the North Sea, however, concluded that discards influence benthic scavenger population dynamics substantially, providing up to 21% of their annual energetic requirements (Catchpole *et al.*, 2006). Both studies stress the need to better understand the underlying causes of these differences and the role of discards in benthic food webs.

Scavenging seabirds are the first in taking advantage of fishery discards and their selective consumption determines the composition and amount of food remaining for others. The share that seabirds are taking is generally large, but varies across regions and seasons at various scales (Garthe *et al.*, 1996). The objective of this study is to partition the discards that are thrown overboard into the share taken up by seabirds, and the remainder which is being returned to the water. While this approach was evaluated for vast areas such as the entire North Sea, local effects have generally been ignored or were not put into a larger perspective (Catchpole *et al.*, 2006; Furness *et al.*, 2007). However, discarding patterns are known to exhibit a wide variability in time and space at various scales (Uhlmann *et al.*, 2013b). Bird abundances also vary across time and space (Certain *et al.*, 2007), as well as their relationship to fisheries at various resolutions (Cama *et al.*, 2012; Louzao *et al.*, 2011). The number of seabirds and the species composition of the flock that is scavenging at fishing vessels are therefore highly variable in space and time (Bartumeus *et al.*, 2010). The numerical abundance and heterospecific interactions of scavenging seabirds at fishing vessels invoke intra- and inter-specific competition, which results in large differences in discard consumption (Camphuysen & Garthe, 1997). This study develops a framework to explicitly account for the spatial and temporal variability in bird presence and composition and to examine whether the variation results in differences in discard consumption. The spatial and temporal variability in discard consumption is then combined with spatial and temporal variation in fisheries' discards to calculate the amount and composition of discards that become available to seabirds or marine scavengers in the water column or on the seabed.

This approach is applied to the Bay of Biscay. The area is characterised by a high diversity in benthic and pelagic fish assemblages, fishing fleets as well as scavenging seabirds (Lorance *et al.*, 2009). We focused on the French fisheries, which include a variety of fleets: bottom trawlers, Nephrops trawlers, gill netters, longliners and pelagic fisheries, targeting a variety of species (from crustaceans over cephalopods to demersal and pelagic fish). The fleets contribute to most of discards in this area and are well documented by year-round monitoring of all discarded species (Cornou *et al.*, 2013; Dubé *et al.*, 2012; Fauconnet *et al.*, 2011). Seabird distributions were also well documented with biannual ship-based and aerial surveys over the full spatial range of the Bay of Biscay (Certain *et al.*, 2007; Pettex *et al.*, 2014).

8.3 Materials and methods

The consumption of fisheries' discards by scavenging seabirds intrinsically depends on a range of seabird and fishery related processes (Bartumeus *et al.*, 2010; Bodey *et al.*, 2014a; Furness *et al.*, 2007). Three major steps can be distinguished within the research strategy. First, the Bay of Biscay case study data related to seabird and fishery processes are presented. Then a framework was defined that combines those processes through a stepwise algorithm. Next, the application of the framework to the case study data was shown to partition discards in the Bay of Biscay, and last the emerging spatial patterns are analysed.

8.3.1 *The Bay of Biscay case study data*

8.3.1.1 *Scavenging seabirds and ship followers*

Aerial and ship-based surveys covered the continental shelf of the Bay of Biscay (ICES Divisions VIIIa and VIIIb). Aerial monitoring was conducted during the first (May-July) and second quarter (December-February) of 2012 and took place during daylight by two observers. Flight height was about 180 m above sea level at a speed of 90 knots. Visual census was accomplished in conditions of limited wave heights and wind speed < 4 Beaufort. Ship-based surveys were realised biannually (April-June; October-November) aboard the RV 'Thalassa' in 2009-2011 following the protocol outlined in Certain *et al.* (2011). Ship followers, defined as seabirds which are effectively attracted to fishing vessels to scavenge upon discards, were also recorded during ship-based surveys. Bottom trawl surveys took place in October and November (second quarter), deploying a 36/47 GOV bottom trawl during 30 min hauls at 4 knots (see ICES (2010) for further details). Trawling during pelagic surveys was conducted between April and June (first quarter) with a pelagic trawl of 40 (horizontal) by 20 m (vertical). Hauls also lasted 30 min at a speed of 4 knots (Certain *et al.*, 2011). Ship followers

were exclusively registered during daylight if they were within a circumference of 200 m. Numbers were recorded by species within five hours after hauling, resulting in 88 observations of ship followers during the first semester and 212 during the second. The number of hauls preceding registration was 1-2 (first semester) or 1-4 (second semester).

8.3.1.2 *Experimental discard consumption by ship followers*

Discard consumption by scavenging ship followers was examined during the bottom trawl survey aboard the RV 'Thalassa' (4-17 November 2013). Fishing took place between 46° and 50° N and 4° and 11° W, and followed specifications outlined in the ICES protocol (ICES, 2010). Sixty-nine standardised discard samples were prepared out of 41 hauls. Each sample contained a mixture of 75 roundfish (Total Length, TL: 9-31cm) and 50 items of another discard type: cephalopods (N=22, mantle length: 3-18cm), *Nephrops norvegicus* (hereafter called *Nephrops*) (N=22, carapace length: 1.8-4.4cm) or boarfish (*Capros aper*) (N=25, TL: 9-17cm) (see species list for the discard types in Table A8.1). *Nephrops* was used as a proxy for benthic invertebrates, given the importance of *Nephrops* trawling. Experiments took place immediately after hauling the gear, and consisted of randomly returning discard items to the sea over a five minutes interval. Bird species and age and the discard type of each successful capture (roundfish, cephalopod, *Nephrops* or boarfish) were recorded. Bird species composition of the flock of ship following seabirds was voice-recorded prior and immediately after each discarded sample.

8.3.1.3 *Discard sampling*

Fishery-dependent data of the French fishing fleet have been collected in the Ifremer onboard observer programme 'Obsmer' to fulfil data requirements of the European Commission Data Collection Framework (EC, 2008a; 2008b). Catch sampling was stratified by metier and quarter. Metiers were defined by the European level 5 definition (EC, 2008a: 57-59), based on gear type, fishing area and target species assemblage. Landed and discarded numbers of each taxon were sampled by fishing operation. Subsamples were raised to the level of fishing operation, and then to trip on the basis of sampled fractions (Fauconnet *et al.*, 2011; Dubé *et al.*, 2012; Cornou *et al.*, 2013). This study focuses on discard and landing data between 2009-2011 in ICES Divisions VIIIa and VIIIb. The six metiers with the highest discarded amounts were selected: (i) bottom trawls targeting demersal fish and cephalopods (hereafter called 'demersal trawlers'), (ii) bottom trawls targeting crustaceans ('*Nephrops* trawlers'), (iii) midwater trawls targeting small pelagic fish ('pelagic trawlers'), (iv) midwater trawl targeting demersal fish and cephalopods ('midwater trawlers'), (v) gill nets and (vi) longlines.

8.3.2 Stepwise framework to estimate the fate of discards

The successive steps of the framework are presented in Figure 8.1. In the first step, the density of scavenging seabirds is estimated by location i and period j . Scavenging seabirds were defined as seabird taxa which are frequently associated with fishing vessels and identified to consume discards. In the second step, the number of ship followers is estimated. The number of ship followers ($F_{i,j,k}$) is estimated for each bird taxon in each location and period and is based on the local bird densities (step 1), their attraction to fishing vessels, and the local number of fishing vessels (equation [1]).

$$F_{i,j,k} = \frac{S_{j,k} * B_{i,j,k}}{V_{i,j}} \quad [1]$$

where $B_{i,j,k}$ is the bird density of taxon k in location i and period j , $V_{i,j}$ the number of fishing vessels in location i and period j and $S_{j,k}$ is the scavenging index of bird taxon k in period j . The scavenging index ($S_{j,k}$) is a time-specific measure that expresses the area over which of a seabird taxon k is attracted to fishing vessels rather than to natural food (Furness *et al.*, 2007) (equation [2]).

$$S_{j,k} = \frac{\overline{n_{j,k}}}{B_{j,k}} \quad [2]$$

where $n_{j,k}$ is the mean number of ship followers and $B_{j,k}$ the mean density of bird taxon k in period j . The scavenging index is related to the radius over which a bird conducts its area-restricted search behaviour (ARS), i.e. increase in turning rate and speed reduction in response to an elevated reward (Fauchald, 2009). The ARS radius was calculated as $\sqrt{(S/\pi)}$, assuming equal attraction from all directions around the vessel (Skov and Durinck, 2001).

The third step in our algorithm relates the number of ship followers to the consumption of discards at experimental level. Experimental Discard Consumption (EDC) was defined as the ratio of the number of discards swallowed ($n_{swallowed}$) to the total number of discarded items thrown (n_{thrown}) (Hudson and Furness, 1989; Camphuysen *et al.*, 1995):

$$EDC_{k,l} = \left[\frac{n_{swallowed_l}}{n_{thrown}} \right]_k \quad [3]$$

EDC was assessed by seabird taxon k and discarded taxon l , following previous approaches (Furness *et al.* 2007).

In the fourth step the total number of discards is estimated and combined with EDC to obtain an estimate of discard consumption at fleet level. Discard consumption is estimated for each framework entity, defined by location i , period j , seabird taxon k and discarded taxon l (Equation 4).

$$C_{i,j,k,l} = \frac{D_{i,j,k}}{d_k} * c_{k,l} \tag{4}$$

where C and c are discard consumption, and D and d are the total number of discards at fleet (capital) and experimental (small letters) level. The total number of discarded items available to marine scavengers other than seabirds is subsequently calculated from Equation 5.

$$R_{i,j,l} = [\sum_k D_k - \sum_k \sum_l C_{k,l}]_{i,j} \tag{5}$$

where R is the remaining fraction of discards of discarded taxon k , available to marine scavengers in the water in location i and period j .

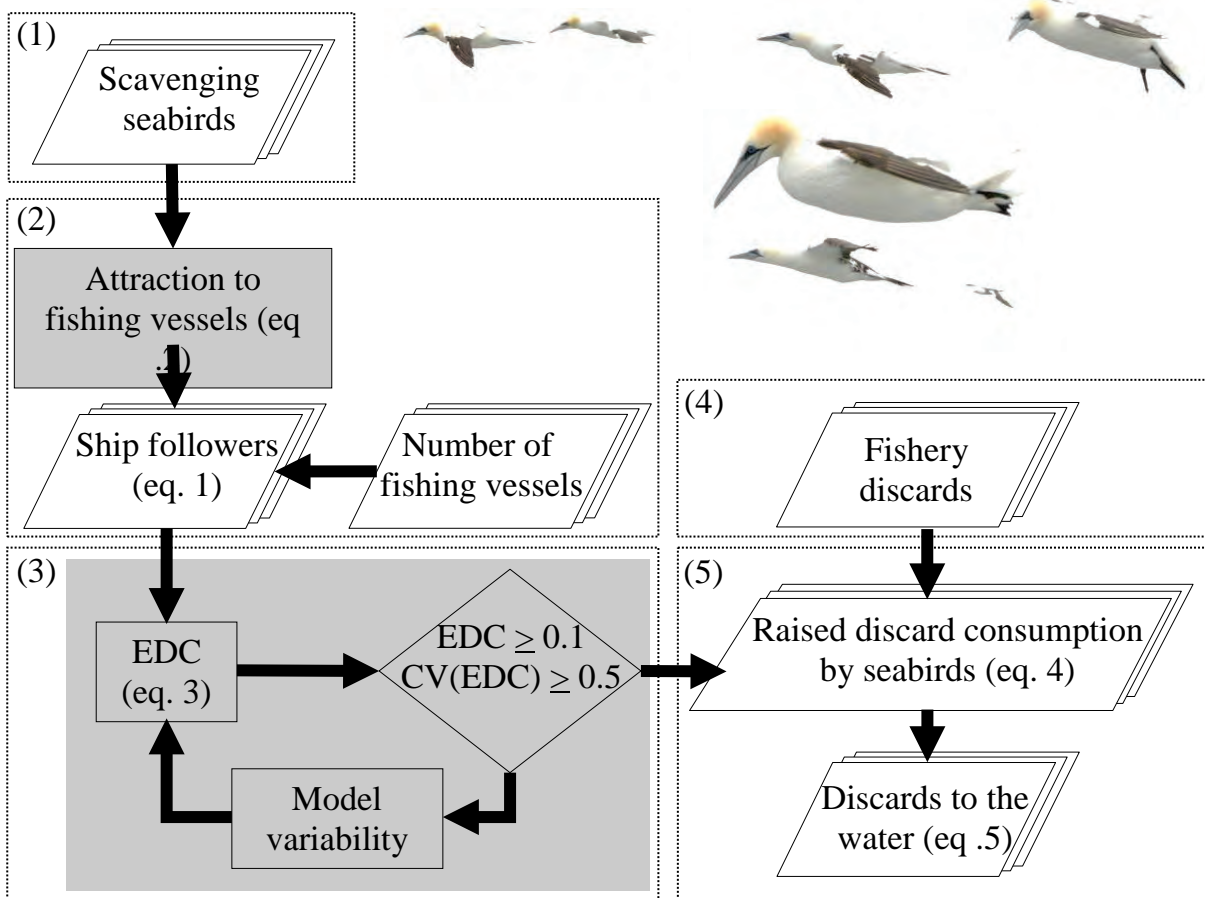


Figure 8.1 Stepwise framework to estimate the fate and consumption of discards in each spatio-temporal entity. Steps (1) to (5) and equations (eq.) are described in this chapter. Parallelograms present spatio-temporal input or output estimates, the rhombus refers to a decision process and rectangles to processing steps. Grey shaded areas were estimated at the scale of the Bay of Biscay: (a) attraction to fishing vessels and (b) Experimental Discard Consumption (EDC). When EDC was ≥ 0.1 and its Coefficient of Variation (CV) was high ≥ 0.5 , the intra- and interguild competition was modelled to explain the variation.

8.3.3 *Applying the framework to the Bay of Biscay case study*

8.3.3.1 *Discretizing framework entities*

Applying the framework required discretization of the framework entities to obtain appropriate linkages between the framework steps. Discretization was based on the lowest resolution of any of the linkages between consecutive steps. For instance, if discard consumption was estimated by species and length class, then the fleet-level estimate of discards was also needed by species and length class in order to estimate the discard consumption at fleet level at the same resolution. Discretization was intended to maximise between-group and minimise within-group variability, and was limited by data availability.

The incorporation of spatial and temporal differences in the framework required that the distributions of discards and seabirds were discretised by the lowest resolution of either distribution. Discards limited inferences in space, while the highest temporal resolution was determined by the biannual monitoring of seabirds. Discards were then standardised by pooling taxa into five discard types, which were defined upon morphological similarities (Camphuysen *et al.*, 1995): benthic invertebrates, cephalopods, depressiform fishes, flatfish and roundfish (Table A8.1, Nikolsky, 1963). Scavenger taxa were pooled into eight foraging guilds upon similar morphology and discard foraging behaviour (Bicknell *et al.*, 2013; Bodey *et al.*, 2014a): Gannets (Sulidae), large gulls, small gulls and terns (henceforth called small gulls), unidentified gulls, Kittiwakes (*Rissa* sp.), Procellariids (Procellariidae), Stormpetrels (Hydrobatidae) and Skuas (Stercorariidae) (see species list in Table A8.2). These guilds included all scavenging seabird species in the Bay of Biscay, except auks (Alcidae) and cormorants (Phalacrocoracidae), which do attend fishing vessels but infrequently use fishery discards (Valeiras, 2003; Bicknell *et al.*, 2013). Unidentified gulls in a spatio-temporal entity were attributed to either small or large gulls following the ratio of local densities of small to large gulls.

8.3.3.2 *Scavenging seabirds and ship followers (framework steps 1 and 2)*

Both aerial and ship-based data (pooled over the years) were processed following the strip transect methodology, assuming that all species were recorded within a strip width of 200 m (aerial, Certain *et al.*, 2008) or 300 m (ship-based, Tasker *et al.*, 1984). The densities of foraging guilds were estimated by ICES Statistical Rectangle (0.5° latitude by 1.0° longitude) to match the spatial distribution of discards. Density calculations were iterated 999 times using random resampling with replacement of bird observations to obtain the coefficient of variation (CV). Density estimates of the aerial and ship-based surveys were compared by calculating the log ratio of the densities in each

rectangle and subsequently smoothing the log ratios with a two-dimensional spline, assuming normal errors and identity link. The fitted values were used to test whether the log ratio in each rectangle differed from zero (Fraser *et al.*, 2008). Significant differences occurred for all foraging guilds. Although both methodologies are common for bird census (Katsanevakis *et al.*, 2012), the advantages of aerial surveys outweighed ship-based surveys for the purpose of this study, especially because scavenging seabirds are more prone to attraction bias by ship-based platforms (Cama *et al.*, 2012). The biannual scavenging index and the number of ship followers in contrast were based on the ship-based survey. The attraction of a specific seabird taxon was assumed to be similar at all locations in the Bay of Biscay and across fishing metiers, because the registrations of the number of ship followers were insufficient to estimate rectangle-specific scavenging indexes. To calculate the number of ship followers, the spatial distribution of fishing vessels was assumed homogenous within each rectangle.

8.3.3.3 Experimental Discard Consumption by ship followers (framework step 3)

EDC was calculated from the Bay of Biscay experiments. EDC estimates from the North Sea were used when Bay of Biscay estimates were absent (EDC of small gulls, flatfish and depressiform fishes, Table 8.1). When EDC and its variability were low for a particular foraging guild and discard type, we assumed that discard consumption in a spatio-temporal entity was only limited by its presence as a ship follower. In contrast, when EDC and its variability were high (EDC > 0.1; CV > 0.5), the intra- and inter-guild competition between ship followers was examined (framework step 3; Figure 8.1). Inter-guild competition was divided into exploitation and interference competition. Exploitation competition does not hamper accessibility to the resources, whereas interference competition reduces in particular this accessibility (Case and Gilpin 1974). Gannets were considered as interference competitors due to their socially dominant behaviour (Hudson and Furness, 1989). EDC was estimated by fitting a logistic regression curve to the predictor variables ‘overall flock size’, ‘number of birds of each foraging guild’ or its natural logarithm, the ‘proportion of the scavenging birds in the flock of interference competitors’ and ‘in the flock of exploitation competitors’ (Lloyd *et al.*, 1967) (Equation 1).

$$EDC = \log_e \left(\frac{p}{1-p} \right) = \beta_0 + \beta_i X_i \quad [6]$$

where β_0 is the intercept and β_i the coefficients for the predictor variables X_i . Logistic regression was based on a Generalised Linear Model (GLM) with logit-link function and quasi-binomial error distribution to account for overdispersion. Collinearity between explanatory variables was examined using a variance inflating factor of two, while influential observations were removed

using the Cook's distance. Models with a lower quasi-Akaike Information Criterion (QAIC) were selected if the $\Delta QAIC$ was >3 . Models with $\Delta QAIC$ of <3 were deemed equal, and the most parsimonious model was selected.

For guilds with low EDC variability during the Bay of Biscay experiments, but which occurred in large flocks in the Bay of Biscay, we used parallel investigations from two data sources in the North Sea. The first series of experiments were conducted aboard the RV 'Belgica' in the southern North Sea (between 52° and 51°N; 1° and 2° E) in December 2011, February, April and December 2012, and April 2013. Gear and fishing specifications followed the outline described in Chapter 5. The experimental protocol largely followed the procedure of the Bay of Biscay experiments, except for the discard samples. Samples contained either 105 (December 2011) or 150 discarded items, composed by two thirds of soles (*Solea solea*; TL: 6-28cm) and one third of roundfish (*Merlangius merlangius* or *Trisopterus* sp.; TL: 9-31cm). A total of 150 experiments were realised. Depressiform fishes were also examined (Rajidae, N=52, TL: 30-163cm) by returning them to sea as a single item during six experiments in December 2011. The second data source from the North Sea was obtained from Camphuysen *et al.* (1995). Pooling the experiments from the North Sea and the Bay of Biscay allowed assessing roundfish consumption by large gulls following the regression procedure outlined above (equation [6]).

Table 8.1 Data sources for estimates of experimental discard consumption by foraging guild and discard type. EDC estimates for roundfish were modelled for large gulls and Gannets (black bold rectangle). EDC-estimates of cephalopods, benthic invertebrates and roundfish were based on experiments in the Bay of Biscay (BoB), except for small gulls consuming roundfish (North Sea). Flatfish estimates were based on experiments in the North Sea (NS), while EDC-estimates of depressiform fishes were based on experiments from various regions. BoB: Bay of Biscay; IVc: southern North Sea (hatched), NS: North Sea (grey).

	Cephalopods	Benthic invertebrates	Roundfish	Flatfish	Depressiform fishes
Small gulls	BoB*	BoB*	NS	NS	BoB*
Procellariids	BoB	BoB	BoB	NS	NS**
Skuas	BoB	BoB	BoB	NS	NS**
Kittiwakes	BoB	BoB	BoB	NS	IVc
Large gulls	BoB	BoB	IVc / NS	NS	IVc
Gannets	BoB	BoB	BoB	NS	NS*

*Assumed negligible; **approximated by other foraging guilds

8.3.3.4 Fisheries' discards (framework step 4)

Total trip discards by discard type were raised to total fleet discards for each of the six métiers in the spatio-temporal entities defined below. Extrapolation was based on a ratio estimator with fishing effort in days at sea as auxiliary variable. Temporal strata were pooled into two periods to match seabird distribution data: April to September ('first semester') and October to March ('second

semester'). Spatial resolution was defined by merging rectangles to enable sufficient discard samples per spatial entity. Rectangles were merged if they occurred in each other's vicinity and reported landings were similar (based on visual inspection of histograms). Each entity was required to include a minimum of ten fishing operations from at least three trips because inter-trip variability is generally larger than within-trip variability (Rochet *et al.*, 2002). Mean sampling coverage of the fishing days in the spatial entities was 1.1% in the first semester and 0.5% in autumn for all métiers (Table A8.3). This was comparable to the mean sampling coverage of the study area without the spatial segregation (first semester: 0.7%, second semester: 0.4%) and to other discard observer programmes (Chapter 4; Rochet *et al.*, 2002).

8.3.3.5 *The fate of discards (framework step 5)*

Discard consumption by seabirds was applied to the fleet-based discard estimates in each semester and rectangle. Discard consumption was not applied in rectangles beyond the limits of the experimental conditions, i.e. in four rectangles in the second semester, comprising >150 ship following Gannets or >220 large gulls. As such, predictions beyond the boundaries of the statistical EDC-models were avoided. To identify which input variables contributed most to the variability in the output variable, the contribution of each input variable to the overall CV was approximated with the Taylor expansion, using the Delta method described in Stratoudakis (1999).

8.3.4 *Spatial pattern analysis*

Conventional statistical parameters (mean, SD, CV) were used to quantify the variability of each framework step. These statistics, however, provide little detail on the spatial organisation of the entities and the resulting spatial pattern in discard partitioning. The Moran's I index was used for spatial pattern analysis (Fortin and Dale, 2005). This index ranges from clustered to dispersed patterns (+1 to -1). Moran's I correlograms were calculated with a lag size of 110 km for each step in our framework (Borcard *et al.*, 2011). The correlograms were categorized into three basic profiles depending on the significance of the Moran's I: (i) autocorrelation only in the smallest distance classes ('patchy' distribution), (ii) positive autocorrelation in short distance classes coupled with negative values in large distance classes ('linear gradient') and (iii) no significant Moran's I coefficients ('random') (Diniz-Filho *et al.*, 2003). To better understand the emerging pattern in the ship followers, the overlap coefficient of Horn (1966) was calculated for bird densities and fishing vessels:

$$Overlap = 2 \frac{p_B p_V}{p_B^2 + p_V^2} \quad [7]$$

where p is the proportional presence of birds (p_B) and the number of fishing vessels (p_V) in location i , and period j as compared to their maximum values in the Bay of Biscay. This coefficient indicated an exact overlap of birds and fishing vessels at a value of 1, and a complete absence of overlap at zero.

8.4 Results

8.4.1 *Estimates of the framework entities*

8.4.1.1 *Scavenging seabirds and ship followers*

The first semester was dominated by high densities of large gulls in the north-eastern and coastal parts of the Bay of Biscay, which were not significantly altered in the second semester ($Z = 1.68$, $P = 0.10$, $r = 0.14$). The densities of Gannets increased significantly in the second semester ($Z = 2.65$, $P < 0.01$, $r = 0.23$) and were mainly located in ICES Division VIIIb. The scavenging index and the radius of ARS behaviour illustrated that large gulls were highly attracted by fishing vessels at all times of the year, while Gannets, Procellariids and Skuas were especially attracted during the second semester (Table 8.2, Table A8.6). The number of ship followers (mean, SD) did not differ significantly between the first (61.5, 163.9) and second semester (70.02, 132.4) ($Z = 0.89$, $P = 0.37$, $r = 0.03$), but its guild composition and spatial organisation did (Table 8.2, Table 8.3, Table A8.6). The first semester was dominated by large gulls with regular occurrence of Gannets and Procellariids in smaller numbers. The flock of ship followers in the second semester was dominated by large gulls and Gannets (>100 individuals).

8.4.1.2 *Experimental Discard Consumption*

Roundfish consumption was higher than the consumption of any other discard type (Figure 8.2, Table A8.4) and varied with flock composition for large gulls and Gannets. Roundfish consumption by Gannets followed a logarithmic increase with the number of ship followers, which explained 76% (Pseudo- R^2) of model variability (Figure 8.3a, Table A8.5). Roundfish consumption by large gulls also followed a logarithmic increase with increasing number of ship followers (Figure 8.3b), but this increase was counteracted by the relative abundance of other competitors (Figure 8.3c, d): large gulls were about three times less effective in capturing discards with increasing relative abundance of Gannets, and up to 0.7 times in competition with other guilds. Both intra- and inter-guild competition explained up to 62% (Pseudo- R^2) of the variability in roundfish consumption by large gulls (Table A8.5).

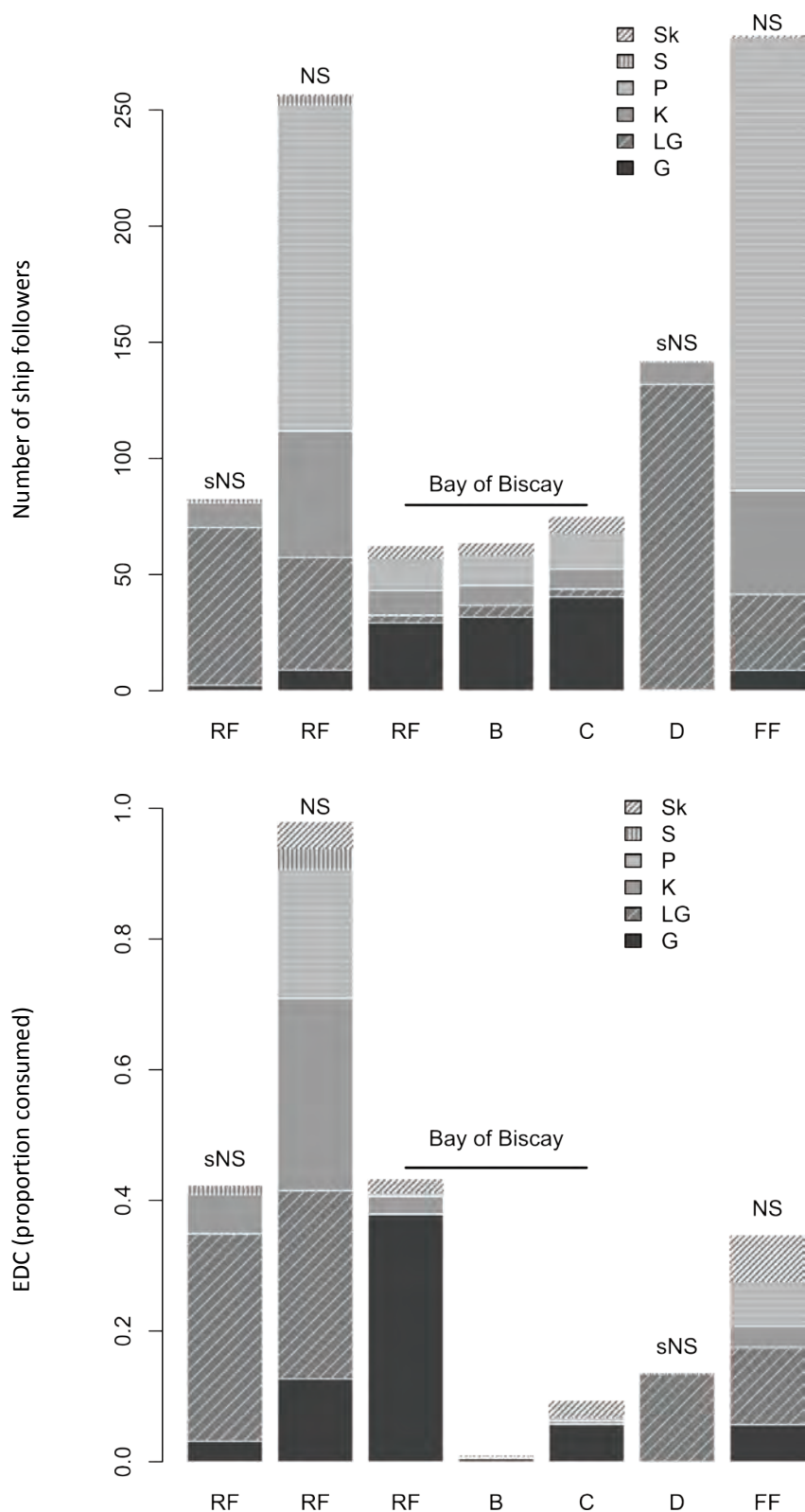


Figure 8.2 Mean number of ship followers (upper) and mean EDC-estimates (lower) of experimental discarding in the Bay of Biscay and the North Sea (sNS: southern North Sea, NS: North Sea). Number of ship followers and composition relate to the experiments in which discard types were discharged. Discard types included Benthic invertebrates (B), cephalopods (C), depressiform fishes (D), flatfish (FF) and roundfish (RF). Foraging guilds included Skuas (Sk), Small gulls (S), Procellariids (P), Kittiwakes (K), Large gulls (LG) and Gannets (G).

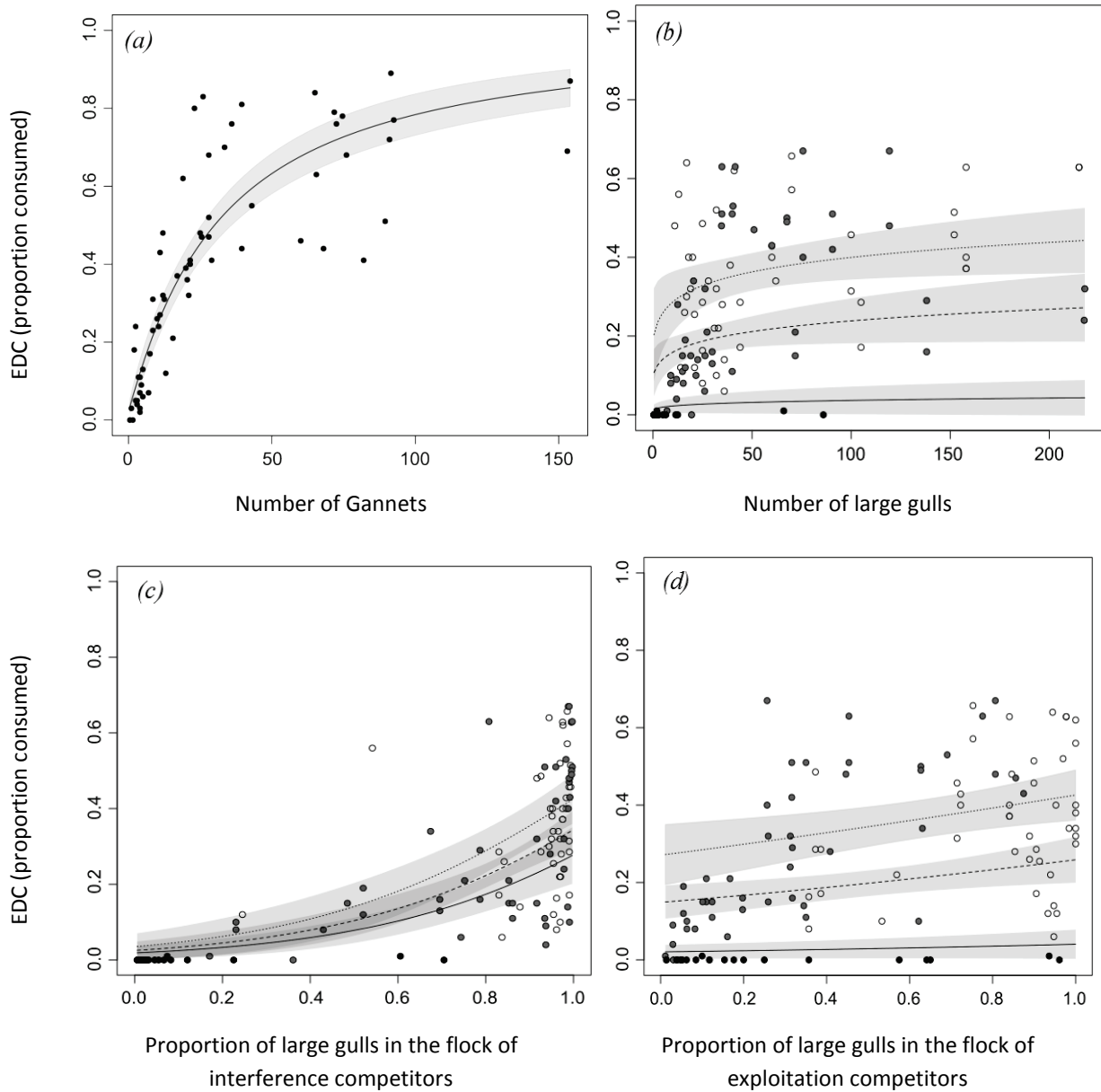


Figure 8.3 Probability of roundfish consumption for Gannets (a) and large gulls (b, c, d). Probabilities are given in function of the number of ship following Gannets (a), ship following large gulls (b), the proportion of interference competitors (c) and the proportion of exploitation competitors (d). The proportion of large gulls in the flock with interference competitors are indicated by a dotted (proportion =1), dashed (proportion=0.75) and solid line (proportion=0.05) in partial plots (b) and (d). The proportion of large gulls in the flock with exploitation competitors are also indicated in panel (c) by a dotted (proportion =1), dashed (proportion=0.5) and solid line (proportion=0.05). Black dots: Bay of Biscay, grey dots: North Sea, open circles: southern North Sea.

8.4.1.3 Fisheries discards

Discarding was mainly concentrated in the north-eastern part of the Bay of Biscay in both semesters with predominantly high numbers of discarded benthic invertebrates and roundfish (Appendix 8.7, Figure 8.4, Figure 11.10). Discarding was particularly apparent during the first semester in the rectangles coinciding with the 'Grande Vasière', a sedimentary mud bank of 12,000 km² which is known as *Nephrops* fishing grounds (24E5-6, 23E5-6, 22E6, 21E7 and 20E8). The benthic discards typically reflect the activity of *Nephrops* trawlers in the Grande Vasière in both semesters. The bulk of the roundfish discards in the first semester was not exclusively caused by *Nephrops* trawlers, but complemented with roundfish discards from pelagic and demersal trawlers in rectangles 21E5-E8 and 21-23E7 (Appendix 8.7, Figure 8.4, Figure 11.10). Roundfish discards by *Nephrops* trawlers in the second semester were concentrated in 23E6 and 24E5, while demersal and pelagic trawlers contributed largely to all other rectangles.

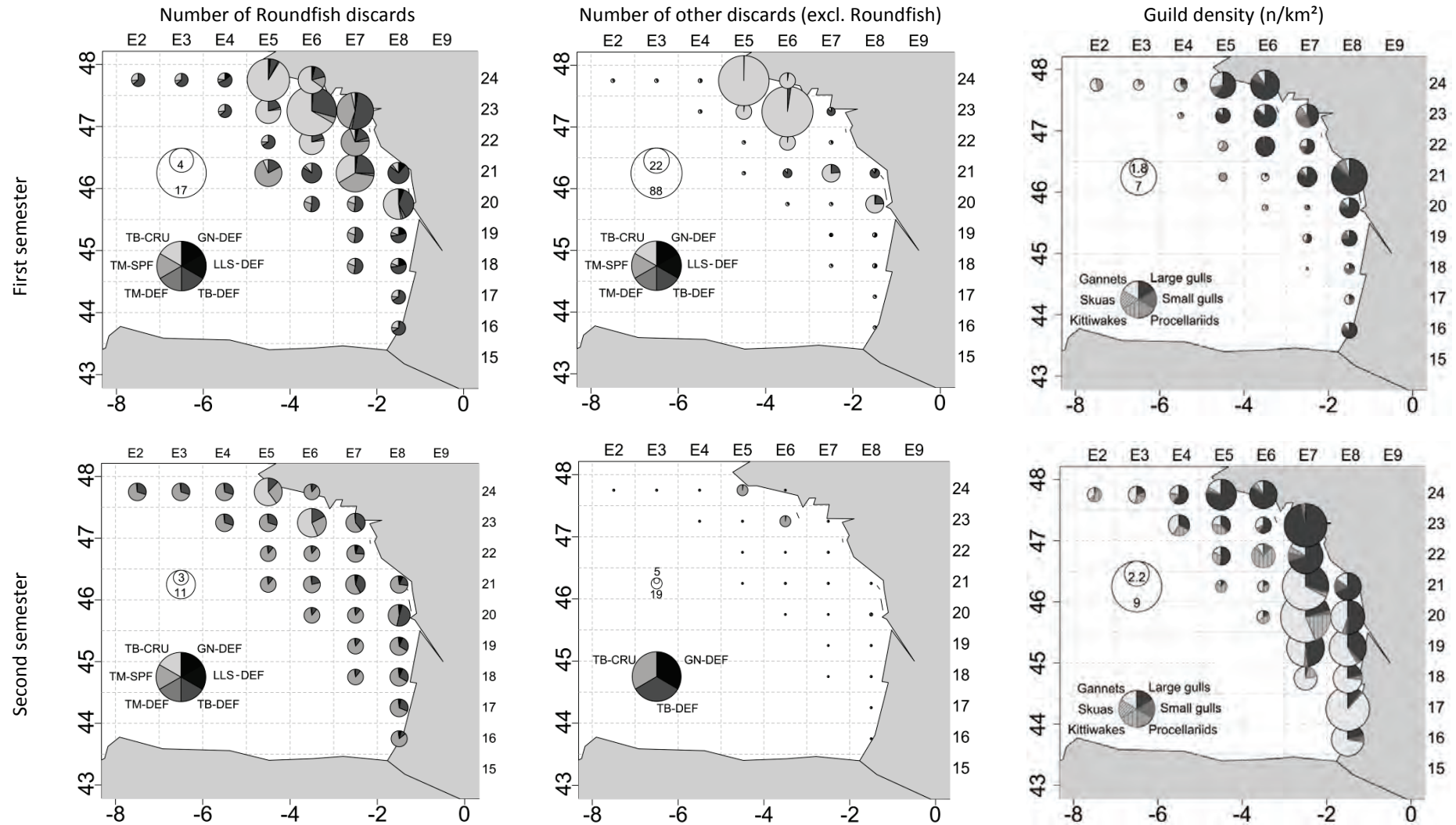


Figure 8.4 Numbers of discarded organisms and seabird densities (n/km^2) during the first (upper panels) and second (lower panels) semester of 2009-2011. Discards are presented by million number of discard items by type: roundfish (left) and all other discard types (middle). Seabird foraging guilds is presented in the right panels.

8.4.2 *Partitioning the fate of discards*

8.4.2.1 *Regional partitioning*

A total of over 500 million items were yearly discarded in the Bay of Biscay, of which 27 % was consumed by seabirds. Two thirds of the discards were produced during the first semester, but consumption by seabirds peaked during the second semester, when they consumed up to half of the discarded material. Both the higher number of discards and the lower discard consumption implied that a significantly higher number of discards became available to marine scavengers in the first semester ($W=1627$, $Z=3.7$, $P<0.001$, $r=0.31$), i.e. 287 million individuals or 77% of the yearly discarded items became available in the first semester. Both semesters showed a high spatial variation. Discard consumption ranged between 7 and 47% of the total number of discarded items across rectangles in the first semester, and between 17 and 85% in the second. Virtually all benthic invertebrates became available to scavengers in the sea, in strong contrast to the discarded roundfish (Figure 8.5, Figure 8.6). The proportion of roundfish consumed (mean, SD) was significantly higher (0.69, 0.20) in the second semester than in the first (0.40, 0.16) ($W=222$, $Z=3.7$, $P<0.0001$, $r=0.32$). This higher consumption was mainly caused by the increased abundance of ship following Gannets, accounting for >50% of the consumption of roundfish in the second semester.

8.4.2.2 *Unexplained variation*

Partitioning the fate of discards by foraging guild and discard type was highly variable across rectangles and semester, as indicated by a mean CV of 4.34 (SD 2.62). Over 50% of the variation was due to the estimates for bird attraction (framework step 2 in Figure 8.1; Table 8.2), while the estimates of EDC (step 3; Table A8.4) and discards (step 4) contributed at least 30% and 10% to the variation, and bird densities less than 5%. The CVs of roundfish consumption by large gulls varied between 1.68 and 2.14 across rectangles. CVs for Gannets varied between 1.97 and 2.17. Variation in roundfish consumption by large gulls and Gannets contributed less to the overall variation in discard partitioning (between 16% and 30% across rectangles and semesters).

8.4.2.3 *Spatial pattern analysis*

The spatial patterns of discard partitioning were predominantly patchy, and reflected the patchy distribution of the fishery discards, except for roundfish (Table 8.3). Roundfish consumption by seabirds imposed different patterns in the amount of discards that became available to scavengers in the sea. The patchy flow of discards in the first semester was only slightly altered, as the number of ship following large gulls were randomly distributed. The random distribution of ship following large gulls resulted from the patchy distribution of large gulls and the patchy distribution of fishing vessels

(Moran's I of 0.22). Fishing vessels were concentrated in coastal regions, whereas large gulls also occurred further away from the coast, which is reflected in a highly variable overlap (Figure 8.7). The spatial pattern of discard consumption in the second semester was driven by a significant linear gradient of Gannets, causing a shift of roundfish consumption towards the south. Gannets were overlapping with fisheries distribution in some rectangles, but also occurred in high abundances in rectangles with few vessels (Figure 8.7).

Table 8.2 Mean (SD, maximum) number of ship followers, mean density, scavenging index and the radius of Area-Restricted Search (ARS) by foraging guild in the Bay of Biscay. The number of hauls, n(hauls), in which the ship followers occurred is indicated with totals of 88 and 212 recordings in the first and second semester respectively.

	Gannets	Large gulls	Small gulls	Kittiwakes	Procel- lariids	Storm- petrels	Skuas
First semester							
N(hauls)	43	72	11	1	31	5	15
N(Ship followers)	4.1 (8.5, 50)	123.7 (233.1, 1506)	4.1 (22.9, 152)	0.3 (3.2, 30)	3.3 (8.5, 60)	0.58 (4.3, 40)	0.72 (2.4, 15)
Density (N /km ²)	0.12	0.21	0.07	0.01	0.06	0.07	0.01
Scavenging index (km ²)	34.2	589.0	58.6	30.0	55.0	8.3	72.0
ARS (km)	3	14	4	3	4	2	5
Second semester							
N (hauls)	175	176	33	62	67	21	124
N (Ship followers)	109.4 (183.5, 1000)	144.0 (179.9, 800)	0.4 (1.1, 1)	5.0 (25.9, 340)	4.9 (20.7, 175)	0.8 (3.9, 42)	3.2 (4.8, 31)
Density (N /km ²)	0.74	0.26	0.04	0.17	0.02	0.04	0.02
Scavenging index (km ²)	147.8	553.8	10.0	29.4	245.0	20.0	160.0
ARS (km)	7	13	2	3	9	3	7

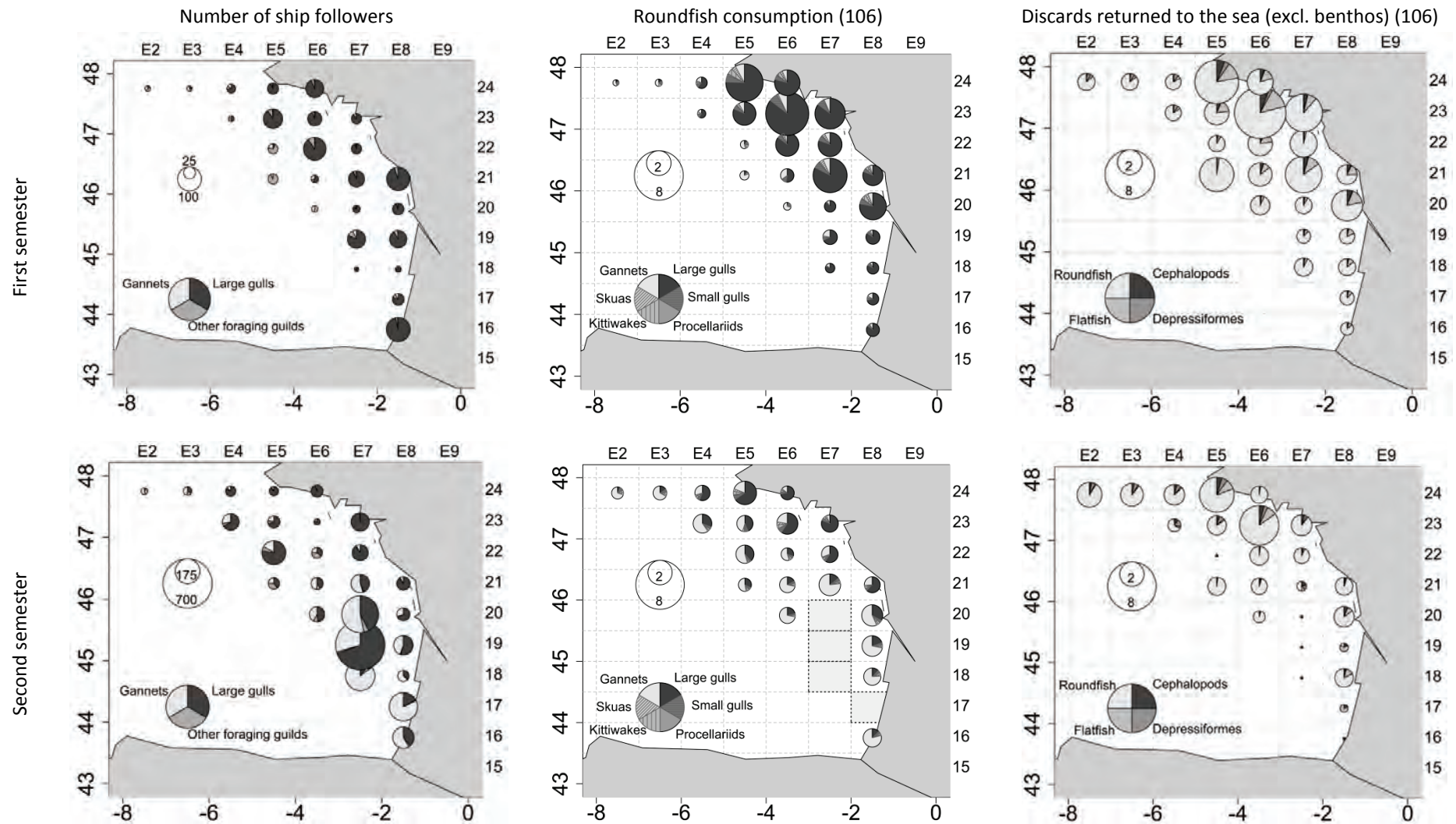


Figure 8.5 Numbers of ship followers, consumption of roundfish by foraging guild (in million numbers) and million number of discards available to other marine scavengers during the first and second semester of 2009-2011. Predictions of roundfish consumption outside model boundaries were disregarded (dotted rectangles)

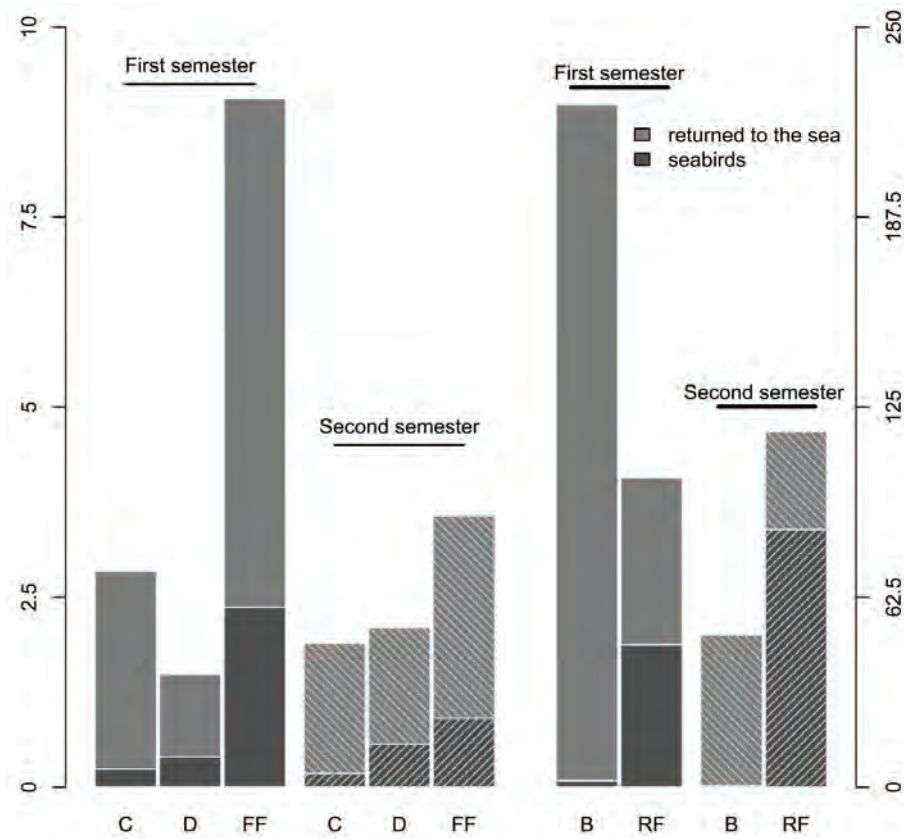


Figure 8.6 Fate of discards in the Bay of Biscay between seabirds and being returned to the sea. Y-axis are expressed in million number of discards. The total numbers of cephalopod (C), depressiform fishes (D) and flatfish (FF) discards respond to the primary axis; benthic invertebrates (B) and roundfish (RF) to the secondary axis. Hatched bars refer to the second semester. Discarded biomass was presented in Appendix 11.3.

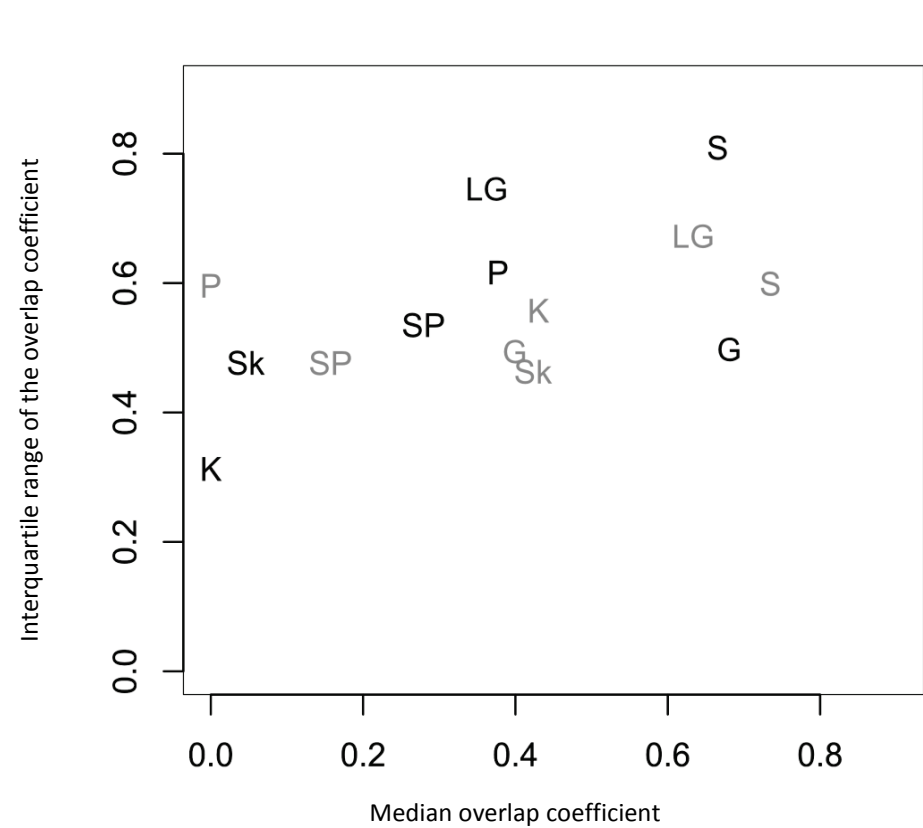


Figure 8.7 Overlap coefficient of bird densities and number of fishing vessels by ICES Rectangles in the Bay of Biscay. Labels indicate the foraging guilds in the first (black) and second (grey) semester: Gannets (G), Large gulls (LG), Small gulls (S), Kittiwakes (K), Skuas (Sk), Procellariids (P) and Stormpetrels (SP). Note the high variability of overlap between ICES Rectangles.

Table 8.3 Categorization of spatial patterns in the correlograms: linear gradient (black cells), patchy (grey cells) or random pattern (white cells). Patchy patterns are indicated by significant Moran's I values at short distances (110 km). Linear gradients are indicated by their highest and lowest Moran's I values at short and long distance respectively.

Semester	Scavenging seabirds		Ship followers		Semester	Fishery discards		Fate of discards	
	First	Second	First	Second		First	Second	First	Second
Gannets	0.28*	0.35* -0.49*	0.09	0.35** -0.45*	Roundfish	0.27**	0.05	0.20*	0.28**
Large gulls	0.21*	0.19*	-0.01	-0.02	Flatfish	0.23*	0.00	0.22*	0.00
Small gulls	0.22*	0.21	-0.08	0.01	Depressiform fishes	0.31**	0.47***	0.31**	0.47***
Kittiwakes	0.25*	-0.12	0.26**	0.08	Cephalopods	0.20*	0.11	0.20*	0.11
Procellariids	-0.04	0.52*** -0.13***	0.18*	0.38***	Benthic invertebrates	0.12	0.08	0.12	0.08
Stormpetrels	0.29**	0.46***	0.19	0.05					
Skuas	0.07	0.13	0.17	0.10					

Significance levels: * < 0.05, ** < 0.01, *** < 0.001

8.5 Discussion

One quarter of the discards in the Bay of Biscay were consumed by seabirds, with a wide variability across foraging guilds, discard types, semesters and locations. The proposed framework accounts for temporal and within-region spatial variability, and thereby increases our ability to estimate the order of magnitude by which discard partitioning changes energy transfer through the food web (Catchpole *et al.*, 2006; Kaiser and Hiddink, 2007). Despite these improvements though, our estimates of the fate of discards remain imprecise. The main causes for this unexplained variation and recommended solutions are discussed in the following Sections. The last Section will demonstrate why a modelling framework is needed and illustrates how spatial scale increasingly gains importance when looking at food web fluxes from birds to benthic invertebrates.

8.5.1 *A modelling framework for discard partitioning*

Spatial and temporal variation of discard consumption by seabirds has been experimentally demonstrated in the nineties by conducting sea trials in different seasons and areas in the North Sea (Garthe *et al.*, 1996). A comparable, large-scale programme has not been undertaken ever since. This study overcomes the logistic and financial demands of such a large-scale programme by presenting a modelling framework as an alternative. The spatial and temporal variation of our results is by consequence conditional on the framework assumptions, in particular those related to (i) the approximations to estimate the number of ship followers in each spatio-temporal entity and (ii) the detection of variation in EDC. The potential sources of unexplained variation and methodological uncertainty are discussed for these two aspects, and solutions are suggested to deal with them.

8.5.1.1 *Scavenging seabirds and ship followers*

The high variability in overlap between birds and fishing vessels indicates the complexity of the relationship between fishing vessels and seabirds at rectangle scale and hence complicates estimations of the number of ship followers. Indeed, the association between birds and fishing vessels is not fully understood in the Bay of Biscay (Certain *et al.*, 2011), and it was not until recently that the influence of fishing vessels on seabird movement patterns has shown clear relationships elsewhere (Bartumeus *et al.*, 2010). It is not surprising that the unexplained variation in the estimates of the number of ship followers was highest amongst all framework steps. Indeed, variation in ship follower estimates was also high in previous studies in the Bay of Biscay (Valeiras, 2003) and elsewhere (Louzao *et al.*, 2011). Data constraints limited the calculations to biannual estimates of the number of ship followers without accounting for spatial variation in bird attraction. The higher number of ship following Gannets in the second semester was likely due to winter migration of Gannets into the Bay of Biscay (Kubetzki *et al.*, 2009) and an increased attraction to

fishery waste during winter (Grémillet *et al.*, 2008). While a seasonal pattern in seabird-fishery interactions was clearly detected, this was not the case for the complex spatial coupling between seabirds and fishing vessels (Louzao *et al.*, 2011). The unexplained variation in the number of ship followers, however, is hypothesized to be largely due to spatial variation in attraction, requiring a rectangle-specific scavenging index.

Differences in seabird attraction across rectangles were not available. Obtaining spatially resolved attraction estimates requires a higher sampling effort across rectangles, or specific resource-demanding experimental designs for each rectangle (Skov and Durinck, 2001), or specific predictive models (Grünbaum and Veit, 2003). The assumption of equal attraction of birds to fishing vessels across all rectangles ignored, however, the foraging strategy of birds and may therefore be a plausible explanation to the high variation in the number of ship followers registered in this study as well as others. A bird's foraging strategy is hierarchically structured in its foraging range and areas of ARS behaviour nested within this foraging range (Fauchald, 2009). The locations and characteristics of the foraging range are strongly influenced by the physical oceanography and its linkage with natural prey (50 km in the Bay of Biscay, Certain *et al.*, 2011). The spatial extent of most seabird foraging ranges is at least 72-200 km (Pettex *et al.*, 2010; Votier *et al.*, 2011) and falls within the scale at which behavioural responses (e.g. competition) differ with the available food sources (300-400 km, Fauchald *et al.*, 2011). The spatial resolution of our framework, i.e. rectangles (approximately 40 x 100km) or aggregations of rectangles, are well suited to reflect this hierarchical level of a bird's foraging behaviour. In contrast, this study did not account for the ARS behaviour within each rectangle. Birds look for conspecifics or indicators of prey patches, such as fishing vessels, to locate their prey indirectly. This process, known as local enhancement, leads to increased foraging success with increasing bird densities and densities of prey or indicators of prey, such as fishing vessels and hydrographic features promoting the availability of natural prey (Skov and Durinck, 2001; Certain *et al.*, 2011). Local enhancement is particularly evident on smaller spatial scales, notably within the foraging range (Fauchald, 2009). The spatial organisation of birds and fisheries within each rectangle therefore invokes rectangle-specific variation in bird attraction to fishing vessels. A thorough understanding of these local processes is required to estimate rectangle-specific bird attraction, and may emerge from recent developments in coupling high-resolution fisheries data with seabird tracking data. Bodey *et al.* (2014b), for instance, revealed that Gannet behaviour is influenced by fishing vessels up to distances of 11 km in the Irish Exclusive Economic Zone. High resolution data will facilitate the estimation of rectangle-specific attraction, which can be readily incorporated in the presented framework, and which will likely reduce the major source of variability in estimating the fate of discards.

8.5.1.2 *Experimental Discard Consumption*

The variation of EDC was high for all discard types and foraging guilds, but largely explained by intra- and inter-guild competition when consumption was higher than a fixed threshold. Indeed, the composition of the ship following flock greatly affects a species' ability to capture discards, depending on their social behaviour and feeding strategy (Camphuysen and Garthe, 1997; Sotillo *et al.*, 2014). The contribution of EDC variation to the overall variation of discard partitioning is, by consequence, mainly caused by the foraging guilds and the discard types of which only few discards were consumed. The unexplained variation of those other discard types (flatfish, etc.) and of guilds other than large gulls and gannets may also be caused by competition.

Intra- and inter-guild competition is suggested as the main driver of the variation reflected in the CVs, since other drivers of EDC variability were controlled for during the sea trials. These drivers comprise the discharge rate and discard composition. The experimental discharge rate was fixed at 1 item per 2.4 seconds during a five minute interval to avoid over-estimation in discard consumption by testing single discarded items (Garthe *et al.*, 1998). This discharge rate reflected discharge rates in gill net or long line fisheries, but may be lower than the pulsed discharges of for instance demersal and pelagic trawling. The discharge rate and the time interval between discarding events significantly affect EDC (Pierre *et al.*, 2010) and may have induced bias in the estimates at fleet level. The size of the discarded items also significantly affects EDC for all discard types and foraging guilds. Gannets for instance prefer roundfish discards ≥ 25 cm, while Kittiwakes can barely swallow them (Garthe *et al.*, 1998). The discard size ranges in the experiments generally matched the size ranges of the fleet discards, e.g. between 7-20 cm for roundfish in pelagic and 8-26 cm in *Nephrops* trawling (Table A8.7). Size-specific EDC was not included in this study owing to a lack of length measurements for all identified discard items (>400 taxa). The high diversity of discarded items also required discretization in discard types (Garthe *et al.*, 1996), and composition. Experimental discard composition consisted of 60% roundfish, which reflected the discard composition of demersal trawlers (approximately 60-70% roundfish discards) and gill netters in the first semester, but less so for *Nephrops* or pelagic trawlers, discarding respectively 17% and 100% of roundfish discards. Roundfish discards consisted of a wide range of morphologies, including gadoids as well as compressiform fish (e.g. boarfish). EDC of compressiform fish differed greatly (Table A8.4), but its effect on the overall fate of the discards was limited in the Bay of Biscay, as compressiform fish contributed <2% of the discards at fleet level.

8.5.1.3 *Recommendations to reduce uncertainty and variability in discard partitioning*

The variability and uncertainties in estimating of the number of ship followers and EDC are further discussed and illustrated in Appendix 11.3. These investigations suggest that the major sources of

uncertainty are linked to metier-specific discarding practices. This study could not cover all metier-specific aspects, and therefore recommends that future experiments take into account the variation of metier-specific discharging procedures as well as discard composition in estimating EDC. The effect of different discharging rates and discard composition may particularly gain momentum when changes occur as a consequence of the EU landing obligation (EU, 2013a).

8.5.2 *Spatial implications of discard partitioning in the food web*

Discarding provides food items to species which would otherwise not have access to them, and creates shortcuts in trophic relationships, leading to higher productivity of the most efficient scavengers (Heath *et al.*, 2014a). Increases in seabird populations are directly affected by discarding. Whether discarding also leads to similar, direct changes for marine scavengers in the sea involves more complex processes. Population changes of benthic scavengers, as an example, depend on the amount of biomass that is extracted by seabirds as their competitive superiors, and on the spatial scale at which this occurs, as benthic scavengers are less mobile than seabirds and therefore less able to move around to access the discards. A tentative example will illustrate that the potential contribution of discards to benthic scavenger's diet varies as one considers decreasing spatial scales.

Scavenging benthic invertebrates represent about 21% of the total benthic biomass in the Grande Vasière and are dominated by *Nephtys caeca*, *Glycera rouxii*, *Natatolana borealis* and *Nephrops* (Le Loc'h *et al.*, 2008). The scavenger biomass crudely equals 60,000 tonnes (Ricciardi & Bourget, 1998; Le Loc'h *et al.*, 2008). Accounting for a yearly consumption rate (Q/B) of 11.2, discards in the Grande Vasière may contribute to 1.6% of the total food requirements of the scavenging benthic community, or 1.0% after seabird predation (Lassalle *et al.*, 2011). These calculations illustrate that the potential of discards as subsidies to the benthic scavenging community in the Grande Vasière is small. Similar findings were found for the entire Bay of Biscay (Lassalle *et al.*, 2011) and the North Sea (1-3% in Groenewold & Fonds, 2000; 7% in Kaiser & Hiddink, 2007).

These findings are contrasted with those of Catchpole *et al.* (2006), suggesting that discards can increase populations of certain benthic scavengers. Their study area was confined to a local fishing ground of 2504 km², which occasionally concentrated >80 vessels at one time. Assuming that all discards in the Grande Vasière are consumed by *Nephrops* showed that discarding can fulfil 14% of their food requirements, taking account of discard consumption by seabirds. *Nephrops* trawling, however, is mainly concentrated in the northern rectangles of the Grande Vasière (24E5, 23E5 and 23E6), resulting in ~80% of the catches. Between 9% and 15% of the food requirements of *Nephrops*

can be provided by discards in those rectangles, using rectangle-specific estimates of *Nephrops* biomass inferred from commercial catches. In contrast, between 12% and 42% of the food requirements of *Nephrops* may be fulfilled by discards in the southern rectangles (20E8, 21E7 and 22E6) where bottom and pelagic trawling also contributed largely to the discards (Figure 8.4 and Figure 8.5).

Unquestionably, several additional factors also affect the potential of discards to subsidize certain components in the benthic food web, such as composition, aggregation and competitive abilities of the scavengers' community (Groenewold and Fonds, 2000), trawling-induced mortalities (Kaiser and Hiddink, 2007) and differences in survival potential between discarded taxa (Chapter 5). Albeit not covering these aspects, discards contribute little to the food requirements of the total scavenging benthic communities when estimated at a large scale (<2% in the Grande Vasière), although they may provide a substantial contribution to the food requirements of certain scavengers at a local scale (up to 42% in certain ICES Rectangles).

8.6 Acknowledgments

The authors would like to thank the Euromarine Mobility Fellowship and the EU-FP7 project BENTHIS (grant no. 312088) for financial support. We are indebted to Verena Trenkel, Laurence Fauconnet, Benoît Dubé, Emeline Pettex, Vincent Ridoux, Olivier Van Canneyt, Alejandro Sotillo, Hans Polet and other colleagues from Ifremer, ILVO, INBO and Pelagis for valuable advice during the development of the approach. We thank the crew of RV 'Belgica' and the RV 'Thalassa' for logistic support during sampling.

8.7 Appendices

Table A8.1 List of taxa included in the discard types. Bold taxa were used in the experimental discarding study in the Bay of Biscay. Categorization was based on morphology which is related to handling time of consumers.

Discard types	Taxa list
Benthic invertebrates	<i>Aequipecten opercularis</i> , <i>Aphroditidae</i> , <i>Asterias rubens</i> , <i>Atelecyclus undecimdentatus</i> , <i>Atrina pectinata</i> , <i>Buccinum undatum</i> , <i>Callinectes</i> sp., <i>Callinectes ornatus</i> , <i>Callista chione</i> , <i>Cancer pagurus</i> , <i>Carcinus maenas</i> , <i>Caridea</i> , <i>Caryophyllia (Caryophyllia) smithii</i> , <i>Cerastoderma edule</i> , <i>Chaceon affinis</i> , <i>Charonia lampas</i> , <i>Chlamys</i> sp., <i>Chlamys islandica</i> , <i>Corystes cassivelaunus</i> , <i>Crangon crangon</i> , <i>Crassostrea gigas</i> , <i>Crepidula fornicata</i> , <i>Crustacea</i> , <i>Dendrophyllia cornigera</i> , <i>Echinoidea</i> , <i>Echinus esculentus</i> , <i>Galathea</i> , <i>Galathea trigosa</i> , <i>Galatheididae</i> , <i>Glycymeris glycymeris</i> , <i>Goneplax rhomboides</i> , <i>Hippocampus</i> sp., <i>Hippocampus hippocampus</i> , <i>Homarus gammarus</i> , <i>Liocarcinus depurator</i> , <i>Liocarcinus navigator</i> , <i>Lutraria lutraria</i> , <i>Macropodia tenuirostris</i> , <i>Maja brachydactyla</i> , <i>Maja squinado</i> , <i>Marsupenaeus japonicus</i> , <i>Mimachlamys varia</i> , <i>Munida intermedia</i> , <i>Munida rugosa</i> , <i>Munnidae</i> , <i>Mytilus</i> sp., <i>Mytilus edulis</i> , <i>Natantia</i> sp., <i>Necora puber</i> , <i>Nephrops norvegicus</i> , <i>Ostrea edulis</i> , <i>Pagurus alatus</i> , <i>Pagurus bernhardus</i> , <i>Palaemon serratus</i> , <i>Palinurus</i> sp., <i>Palinurus elephas</i> , <i>Palinurus mauritanicus</i> , <i>Panulirus laevicauda</i> , <i>Parapenaeus longirostris</i> , <i>Paromola cuvieri</i> , <i>Pecten jacobaeus</i> , <i>Pecten maximus</i> , <i>Polybius henslowii</i> , <i>Portunidae</i> , <i>Portunus</i> sp., <i>Psammechinus miliaris</i> , <i>Pteroeides griseum</i> , <i>Rhizostoma pulmo</i> , <i>Scyllarides delfosi</i> , <i>Scyllarus arctus</i> , <i>Scyphozoa</i> , <i>Solenidae</i> , <i>Squilla mantis</i> , <i>Tritonia hombergii</i>
Cephalopods	<i>Alloteuthis</i> , <i>Alloteuthis media</i> , <i>Cephalopoda</i> , <i>Eledone</i> sp. , <i>Eledone cirrhosa</i> , <i>Eledone moschata</i> , <i>Illex</i> sp. , <i>Illex coindetii</i> , <i>Loligo</i> sp. , <i>Loligo forbesi</i> , <i>Loligo vulgaris</i> , <i>Octopodidae</i> , <i>Octopus</i> sp., <i>Octopus vulgaris</i> , <i>Ommastrephidae</i> , <i>Rossia macrosoma</i> , <i>Sepia</i> sp., <i>Sepia elegans</i> , <i>Sepia officinalis</i> , <i>Sepia orbignyana</i> , <i>Sepiola</i> sp., <i>Sepiola affinis</i> , <i>Sepiola atlantica</i> , <i>Teuthoidea</i> , <i>Todarodes sagittatus</i> , <i>Todaropsis eblanae</i>
Depressiformes (excl. flatfish)	<i>Amblyraja radiata</i> , <i>Dasyatis pastinaca</i> , <i>Dipturus batis</i> , <i>Dipturus oxyrinchus</i> , <i>Leucoraja circularis</i> , <i>Leucoraja fullonica</i> , <i>Leucoraja naevus</i> , <i>Lophiidae</i> , <i>Lophius</i> sp., <i>Lophius budegassa</i> , <i>Lophius piscatorius</i> , <i>Mobula hypostoma</i> , <i>Myliobatis</i> sp., <i>Myliobatis aquila</i> , <i>Raja</i> sp., <i>Raja asterias</i> , <i>Raja brachyura</i> , <i>Raja clavata</i> , <i>Raja microocellata</i> , <i>Raja montagui</i> , <i>Raja undulata</i> , <i>Rajidae</i> , <i>Rhinoptera bonasus</i> , <i>Torpedosp.</i> , <i>Torpedo marmorata</i> , <i>Torpedo nobiliana</i> , <i>Torpedo torpedo</i>
Flatfish	<i>Arnoglossus</i> sp., <i>Arnoglossus imperialis</i> , <i>Arnoglossus laterna</i> , <i>Arnoglossus thori</i> , <i>Buglossidium luteum</i> , <i>Citharus linguatula</i> , <i>Dicologlossa cuneata</i> , <i>Glyptocephalus cynoglossus</i> , <i>Hippoglossoides platessoides</i> , <i>Lepidorhombus</i> sp., <i>Lepidorhombus boscii</i> , <i>Lepidorhombus whiffiagonis</i> , <i>Limanda ferruginea</i> , <i>Limanda limanda</i> , <i>Microchirus</i> sp., <i>Microchirus variegatus</i> , <i>Microstomus kitt</i> , <i>Pegusa lascaris</i> , <i>Phrynorhombus norvegicus</i> , <i>Platichthys flesus</i> , <i>Pleuronectes platessa</i> , <i>Pleuronectiformes</i> , <i>Scophthalmus maximus</i> , <i>Scophthalmus rhombus</i> , <i>Solea</i> sp., <i>Solea senegalensis</i> , <i>Solea solea</i> , <i>Soleidae</i> , <i>Zeugopterus punctatus</i>
Roundfish	<i>Acanthostracion quadricornis</i> , <i>Agonus cataphractus</i> , <i>Alepocephalus</i> , <i>Alepocephalus bairdii</i> , <i>Alepocephalus rostratus</i> , <i>Alopias vulpinus</i> , <i>Alosa alosa</i> , <i>Alosa fallax</i> , <i>Ammodytes</i> , <i>Ammodytes marinus</i> , <i>Ammodytes tobianus</i> , <i>Ammodytidae</i> , <i>Anarhichas</i> , <i>Anguilla anguilla</i> , <i>Aphanopus carbo</i> , <i>Aphia minuta</i> , <i>Apogon nigrocincta</i> , <i>Apogon noumeae</i> , <i>Apristurus</i> , <i>Argentina silus</i> , <i>Argentina sphyraena</i> , <i>Argyrolepecus olfersii</i> , <i>Argyrosomus regius</i> , <i>Aspistor quadriscutis</i> , <i>Atherina presbyter</i> , <i>Auxis rochei rochei</i> , <i>Bagre bagre</i> , <i>Balistes caprisicus</i> , <i>Balistidae</i> , <i>Belone belone</i> , <i>Beryx decadactylus</i> , <i>Beryx splendens</i> , <i>Blennius</i> , <i>Blennius ocellaris</i> , <i>Boops boops</i> , <i>Brama brama</i> , <i>Brosme brosme</i> , <i>Callionymus</i> , <i>Callionymus lyra</i> , <i>Callionymus maculatus</i> , <i>Callionymus reticulatus</i> , <i>Capros aper</i> , <i>Carangidae</i> , <i>Caranx latus</i> , <i>Centrolabrus exoletus</i> , <i>Centrolophus niger</i> , <i>Centrophorus squamosus</i> , <i>Centroscymnus coelolepis</i> , <i>Centroselachus crepidater</i> , <i>Cepola macrophthalma</i> , <i>Cetorhinus maximus</i> , <i>Chelidonichthys</i> , <i>Chelidonichthys cuculus</i> , <i>Chelidonichthys lucernus</i> , <i>Chelidonichthys obscurus</i> , <i>Chelon labrosus</i> , <i>Chimaera monstrosa</i> , <i>Chlorophthalmus agassizi</i> , <i>Chromis chromis</i> , <i>Cichlasoma bimaculatum</i> , <i>Ciliata mustela</i> , <i>Clupea harengus</i> , <i>Coelorinchus caelorhincus</i> , <i>Coelorinchus labiatus</i> , <i>Conger</i> , <i>Conger conger</i> , <i>Coris julis</i> , <i>Coryphaenoides guentheri</i> , <i>Coryphaenoides rupestris</i> , <i>Ctenolabrus rupestris</i>

Table A8.1 (continued) List of taxa included in the discard types.

Discard types	Taxa list
Roundfish	<p><i>Cubiceps gracilis</i>, <i>Cyclopterus lumpus</i>, <i>Dalatias licha</i>, <i>Deania calcea</i>, <i>Dicentrarchus</i>, <i>Dicentrarchus labrax</i>, <i>Dicentrarchus punctatus</i>, <i>Diplodus annularis</i>, <i>Diplodus cervinus</i>, <i>Diplodus puntazzo</i>, <i>Diplodus sargus</i>, <i>Diplodus sargus cadenati</i>, <i>Diplodus vulgaris</i>, <i>Echiichthys vipera</i>, <i>Enchelyopus cimbrius</i>, <i>Engraulis encrasicolus</i>, <i>Entelurus aequoreus</i>, <i>Epigonus telescopus</i>, <i>Epinephelus itajara</i>, <i>Etmopterus princeps</i>, <i>Etmopterus spinax</i>, <i>Euthynnus alletteratus</i>, <i>Eutrigla gurnardus</i>, <i>Gadiculus argenteus</i>, <i>Gadiculus argenteus argenteus</i>, <i>Gadiformes</i>, <i>Gadus morhua</i>, <i>Gaidropsarus macrophthalmus</i>, <i>Gaidropsarus mediterraneus</i>, <i>Gaidropsarus vulgaris</i>, <i>Galeorhinus galeus</i>, <i>Galeus melastomus</i>, <i>Galeus murinus</i>, <i>Ginglymostoma cirratum</i>, <i>Gobiidae</i>, <i>Gobius niger</i>, <i>Gymnammodytes semisquamatus</i>, <i>Halargyreus johnsonii</i>, <i>Helicolenus dactylopterus</i>, <i>Heptranchias perlo</i>, <i>Hexanchus griseus</i>, <i>Hoplostethus atlanticus</i>, <i>Hoplostethus mediterraneus</i>, <i>Hoplostethus mediterraneus mediterraneus</i>, <i>Hydrolagus</i>, <i>Hydrolagus mirabilis</i>, <i>Hygophum benoiti</i>, <i>Hymenocephalus italicus</i>, <i>Hyperoplus lanceolatus</i>, <i>Isurus oxyrinchus</i>, <i>Katsuwonus pelamis</i>, <i>Labridae</i>, <i>Labrus</i>, <i>Labrus bergylta</i>, <i>Labrus mixtus</i>, <i>Lamna nasus</i>, <i>Lampetra fluviatilis</i>, <i>Lepadogaster</i>, <i>Lepidion eques</i>, <i>Lesueurigobius friesii</i>, <i>Lithognathus mormyrus</i>, <i>Liza</i>, <i>Liza aurata</i>, <i>Liza ramada</i>, <i>Liza saliens</i>, <i>Macroramphosus scolopax</i>, <i>Macrourus berglax</i>, <i>Malacocephalus laevis</i>, Melanogrammus aeglefinus, <i>Menticirrhus americanus</i>, <i>Merlangius merlangus</i>, Merluccius merluccius, Micromesistius poutassou, <i>Mola mola</i>, <i>Molva</i>, <i>Molva dypterygia</i>, <i>Molva macrophthalma</i>, <i>Molva molva</i>, <i>Mora moro</i>, <i>Morone saxatilis</i>, <i>Mugil</i>, <i>Mugil cephalus</i>, <i>Mugil curema</i>, <i>Mugilidae</i>, <i>Mullidae</i>, <i>Mullus</i>, <i>Mullus barbatus</i>, <i>Mullus surmuletus</i>, <i>Muraenesocidae</i>, <i>Mustelus</i>, <i>Mustelus asterias</i>, <i>Mustelus mustelus</i>, <i>Mustelus punctulatus</i>, <i>Myoxocephalus scorpioides</i>, <i>Neocyttus helgae</i>, <i>Nezumia aequalis</i>, <i>Osmerus eperlanus</i>, <i>Pagellus</i>, <i>Pagellus acarne</i>, <i>Pagellus bogaraveo</i>, <i>Pagellus erythrinus</i>, <i>Pagrus pagrus</i>, <i>Petromyzon marinus</i>, <i>Phycis blennoides</i>, <i>Phycis phycis</i>, <i>Pollachius pollachius</i>, <i>Pollachius virens</i>, <i>Polyprion americanus</i>, <i>Pomacanthidae</i>, <i>Pomacentridae</i>, <i>Pomatoschistus microps</i>, <i>Pomatoschistus minutus</i>, <i>Prionace glauca</i>, <i>Prionotus</i>, <i>Remora remora</i>, <i>Rhizoprionodon</i>, <i>Salmo salar</i>, <i>Salmo trutta fario</i>, <i>Salmo trutta trutta</i>, <i>Sarda sarda</i>, <i>Sardina pilchardus</i>, <i>Sarpa salpa</i>, <i>Schedophilus medusophagus</i> + <i>S. ovalis</i>, <i>Sciaena umbra</i>, <i>Scomber colias</i>, <i>Scomber japonicus</i>, Scomber scombrus, <i>Scomberesox saurus saurus</i>, <i>Scomberomorus</i>, <i>Scomberomorus brasiliensis</i>, <i>Scomberomorus regalis</i>, <i>Scombridae</i>, <i>Scorpaena</i>, <i>Scorpaena elongata</i>, <i>Scorpaena notata</i>, <i>Scorpaena porcus</i>, <i>Scorpaena scrofa</i>, <i>Scorpaeniformes</i>, <i>Scyliorhinidae</i>, <i>Scyliorhinus</i>, <i>Scyliorhinus canicula</i>, <i>Scyliorhinus stellaris</i>, <i>Scymnodon ringens</i>, Sebastes, <i>Sebastes norvegicus</i>, <i>Sebastidae</i>, <i>Seriola</i>, <i>Seriola carpenteri</i>, <i>Seriola rivoliana</i>, <i>Serranus</i>, <i>Serranus cabrilla</i>, <i>Sparidae</i>, <i>Sparus</i>, <i>Sparus aurata</i>, <i>Spondylisoma cantharus</i>, <i>Sprattus sprattus</i>, <i>Squalidae</i>, <i>Squalus acanthias</i>, <i>Symphodus melops</i>, <i>Symphodus tinca</i>, <i>Synchiropus phaeton</i>, <i>Syngnathidae</i>, <i>Syngnathus acus</i>, <i>Taurulus bubalis</i>, <i>Thunnus</i>, <i>Thunnus alalunga</i>, <i>Thunnus albacares</i>, <i>Thunnus obesus</i>, <i>Thunnus thynnus</i>, <i>Trachinus draco</i>, <i>Trachinus radiatus</i>, <i>Trachipterus arcticus</i>, <i>Trachipterus trachipterus</i>, <i>Trachurus</i>, <i>Trachurus mediterraneus</i>, <i>Trachurus picturatus</i>, Trachurus trachurus, <i>Trachyrincus murrayi</i>, <i>Trachyrincus scabrus</i>, <i>Trachyscorpia cristulata cristulata</i>, <i>Trachyscorpia cristulata echinata</i>, <i>Triakidae</i>, <i>Trichiurus lepturus</i>, <i>Trigla</i>, <i>Trigla lyra</i>, <i>Triglidae</i>, <i>Trigloporus lastoviza</i>, Trisopterus, <i>Trisopterus esmarkii</i>, <i>Trisopterus luscus</i>, <i>Trisopterus minutus</i>, <i>Umbrina canariensis</i>, <i>Umbrina cirrosa</i>, <i>Xiphias gladius</i>, <i>Zenopsis conchifer</i>, <i>Zeus faber</i></p>

Table A8.2 Pooling of seabird scavenging taxa in foraging guilds. Categorization was based on morphology and discard foraging behaviour.

Foraging guilds	Taxa list
<i>Sulidae</i>	<i>Morus bassanus</i>
Large gulls	<i>Larus fuscus</i> , <i>Larus maritimus</i> , <i>Larus argentatus</i> , <i>Marus michahellis</i> , <i>Larus cachinnans</i> , <i>Larus hyperboreus</i>
Small gulls	<i>Larus minutus</i> , <i>Larus melanocephalus</i> , <i>Larus sabini</i> , <i>Larus canus</i> , <i>Larus audouinii</i> , <i>Larus ridibundus</i> , <i>Sterna</i> sp., <i>Sterna albifrons</i> , <i>Sterna hirundo</i> , <i>Sterna paradisaea</i> , <i>Sterna sandvicensis</i> , <i>Sterna dougallii</i>
Unidentified gulls	<i>Larus</i> sp., which could not be classified as large or small gull
<i>Rissa</i> sp.	<i>Rissa tridactyla</i>
<i>Procellariidae</i>	<i>Calonectris</i> sp., <i>Calonectris diomedea</i> , <i>Fulmarus glacialis</i> , <i>Calonectris</i> sp., <i>Puffinus</i> sp., <i>Puffinus gravis</i> , <i>P. griseus</i> , <i>P. yelkouan</i> , <i>P. puffinus</i> , <i>P. mauretanicus</i> , <i>Thalassarche melanophris</i>
<i>Hydrobatidae</i>	<i>Hydrobates</i> sp., <i>Hydrobates pelagicus</i> , <i>Oceanites</i> sp., <i>Oceanodroma</i> sp., <i>Oceanodroma leucorhoa</i>
<i>Stercorariidae</i>	<i>Stercorarius</i> sp., <i>Stercorarius skua</i> , <i>Stercorarius parasiticus</i> , <i>Stercorarius pomarinus</i>

Table A8.3 Data used in this study for the major metiers in the Bay of Biscay. Samples were aggregated across ICES Statistical Rectangles to ensure a sufficient number of samples per spatio-temporal entity. (i) bottom trawls targeting demersal fish and cephalopods (TB-def), (ii) bottom trawls targeting crustaceans (TB-CRU), (iii) midwater trawls targeting small pelagic fish (TM-SPF), (iv) midwater trawl targeting demersal fish and cephalopods (TM-DEF), (v) gill nets sensu latu (GN-DEF) and (vi) longlines (LLS).

	Spatial sites	Sampling characteristics			Reported fleet characteristics		Sampling coverage (%)		Discarded proportion (roundfish)
		Number of hauls (trips)	Fishing days	Landings (kg)	Fishing days	Landings (t)	Fishing days	Landings	
1st semester									
GN-DEF	16E8, 17E8	40 (22)	26	3270	3368	593	0.77	0.55	0.22
	18E8, 19E8, 20E8, 21E8	444 (115)	132	3611	22606	3893	0.58	0.09	0.29
	24E4, 23E6, 21E7, 22E7, 23E7	246 (49)	67	13117	12257	2426	0.55	0.54	0.56
	24E5, 24E6	57 (13)	16	1836	9375	2053	0.17	0.09	0.29
	Remainder*	38 (12)	19	9575	2913	2039	0.65	0.47	0.10
LLS	24E5, 23E5, 24E6, 18E6, 21E6, 22E6, 23E7, 20E8	314 (33)	32	3482	16067	4466	0.20	0.08	0.13
	Remainder*	29 (6)	6	436	5146	2652	0.12	0.02	0.24
	23E5, 22E5, 24E6, 21E7, 20E8	108 (36)	51	20211	13344	5748	0.38	0.35	0.38
TB-CRU	24E5, 23E6	87 (37)	38	12648	21894	7230	0.17	0.17	0.24
	Remainder*	48 (23)	30	9078	6264	2056	0.48	0.44	0.50
TB-DEF	21E6, 21E7, 20E8, 21E8	206 (57)	62	19562	19371	7866	0.32	0.25	0.46
	23E6, 23E7	91 (21)	21	7255	11748	4728	0.18	0.15	0.59
	24E2, 24E3, 23E4, 24E4, 24E5	155 (18)	68	58214	8184	7249	0.18	0.15	0.70
	Remainder*	83 (33)	42	13722	12580	7666	0.33	0.18	0.78
TM-DEF	21E7, 22E7, 23E7, 20E8	30 (12)	20	3314	1515	1423	1.32	0.23	0.93
	24E4, 24E5, 23E6	29 (4)	13	12876	327	516	3.98	2.49	0.10
	Remainder*	10 (5)	9	4968	1184	813	0.76	0.61	0.39
TM-SPF	20E6, 18E6, 19E6, 20E7	18 (9)	8	18365	258	916	3.10	2.01	0.25
	15E8, 16E8, 17E8, 18E8, 19E8, 20E8	25 (13)	13	21213	396	643	3.28	3.30	0.12
	21E5, 21E7, 22E7, 23E7	87 (23)	26	95922	4436	11338	0.59	0.85	0.15
	23E6, 24E6	38 (22)	18	104242	1133	2690	1.59	3.87	0.02
	Remainder*	31 (14)	14	59360	398	1819	3.52	3.26	0.13

*The remainder category includes all other rectangles in ICES Division VIIIa/b not listed for the investigated metier.

Table A8.3 (continued). Data used in this study for the major metiers in the Bay of Biscay.

	Sampling characteristics			Reported fleet characteristics		Sampling coverage (%)		Discarded proportion (roundfish)	
	Spatial sites	Hauls (trip)	Fishing days	Landings (kg)	Fishing days	Landings (t)	Fishing days		Landings
2nd semester									
GN-DEF	16E8, 17E8	69 (33)	31	6132	2960	838	1.05	0.73	0.39
	18E8, 19E8, 20E8, 21E8	453 (114)	140	40353	12226	4239	1.15	0.95	0.40
	22E6, 23E6, 24E6, 20E7, 23E7	198 (49)	44	9537	7992	2723	0.55	0.35	0.09
	21E7, 22E7	20 (6)	12	7792	7567	3708	0.16	0.21	0.16
	Remainder*	23 (12)	14	1905	3534	2189	0.40	0.09	0.27
	LLS	24E4, 24E5, 24E6, 23E6, 23E7, 21E8, 20E8	15 (3)	3	226	7219	2953	0.04	0.01
Remainder*		13 (5)	5	876	4123	2338	0.12	0.04	0.11
TB-CRU	24E5, 23E6	93 (37)	44	10339	12958	3519	0.34	0.29	0.36
	Remainder*	12 (8)	9	1374	5679	1643	0.16	0.08	0.23
TB-DEF	17E8, 18E8, 19E8	102 (18)	37	19231	5012	4740	0.74	0.41	0.58
	20E8	67 (16)	26	2869	5397	2715	0.48	0.11	0.47
	21E6, 21E8, 22E7	29 (13)	13	2892	7683	4669	0.17	0.06	0.55
	23E6, 23E7, 21E7	32 (10)	14	8465	13789	8256	0.10	0.10	0.57
	24E2, 24E3, 24E4, 24E5, 23E4, 23E5	164 (17)	57	57897	18555	9813	0.31	0.59	0.61
	Remainder*	30 (14)	19	8107	7173	5057	0.26	0.16	0.59
TM-DEF	21E7, 22E7, 23E7	18 (10)	10	14381	1781	2038	0.56	0.71	0.05
	Remainder*	33 (9)	18	13820	1551	1443	1.16	0.96	0.15
TM-SPF	All rectangles	26 (9)	9	51403	2025	6625	0.44	0.78	0.44

*The remainder category includes all other rectangles in ICES Division VIIIa/b not listed for the investigated metier.

Table A8.4 Mean number of scavenging seabirds and mean EDC-estimates (SD, maximum) for the Bay of Biscay experiment and experiments in the North Sea (1: experiments on-board RV 'Belgica', 2: Camphuysen *et al.*, 1995). Stormpetrels did not consume any discard item.

	Bay of Biscay experiments				southern North Sea ¹		entire North Sea ²		
	Roundfish	Boarfish	Cephalopods	Norway lobster	Roundfish	Depressiformes	Roundfish	Flatfish	
Number of scavenging seabirds	Gannets	29.4 (34.2, 154)	15.1 (15.1, 76)	40.4 (40.0, 154)	31.8 (36.3, 153)	2.4 (8.7, 63)	0.3 (1.0, 3)	9.1 (11.3, 37.4)	9.0 (10.9, 34.5)
	Large gulls	3.2 (13.7, 86)	0.8 (2.5, 11.5)	3.5 (12.9, 66)	5.0 (18.6, 86)	67.9 (66.2, 275)	131.7 (51.2, 185)	48.5 (49.0, 217.9)	32.7 (29.4, 217.9)
	Small gulls	-	-	-	-	1.1 (1.6, 10)	-	4.0 (5.3, 12.7)	0.1 (-, 0.1)
	Kittiwakes	10.6 (13.5, 48.5)	15.1 (15.1, 48.5)	8.6 (13.1, 45)	8.6 (11.4, 45)	10.9 (14.1, 70)	9.7 (4.9, 20)	54.4 (68.8, 270.8)	44.6 (69.2, 270.8)
	Procellariids	13.9 (20.9, 95.5)	13.0 (10.4, 37.5)	15.6 (26.6, 95.5)	13.2 (22.0, 84.5)	-	-	140.23 (147.3, 553.8)	194.8 (156.4, 553.8)
	Skuas	5.1 (4.4, 24.5)	3.9 (3.5, 12)	6.7 (5.0, 24.5)	4.8 (4.1, 19)	-	-	0.4 (1.0, 4.8)	0.8 (1.3, 4.8)
	Stormpetrels	0.3 (1.0, 8)	-	0.6 (1.6, 8)	0.3 (0.6, 2)	-	-	-	-
	Flock size	62.2 (32.8, 168)	47.8 (16.9, 80)	74.7 (35.8, 168)	63.5 (35.6, 165)	82.3 (67.9, 281)	142.3 (51.6, 195)	254.7 (200.0, 701.8)	295.8 (198.6, 701.8)
	Bay of Biscay experiments				southern North Sea ¹		entire North Sea ²		
	Roundfish	Boarfish	Cephalopods	Norway lobster	Roundfish	Depressiformes	Roundfish	Flatfish	
EDC estimates	Gannets	0.379 (0.278, 0.89)	0.048 (0.091, 0.36)	0.058 (0.090, 0.38)	0.006 (0.017, 0.08)	0.032 (0.067, 0.353)	-	0.127 (0.140, 0.52)	0.057 (0.031, 0.108)
	Large gulls	0.001 (0.003, 0.01)	0.004 (0.012, 0.04)	0	0.002 (0.006, 0.02)	0.317 (0.294, 0.540)	0.135	0.288 (0.226, 0.91)	0.118 (0.147, 0.5)
	Small gulls	-	-	-	-	0.013 (0.017, 0.020)	-	0.032 (0.106, 0.43)	<0.01 (-, 0.01)
	Kittiwakes	0.026 (0.040, 0.17)	0	0.005 (0.012, 0.04)	0.001 (0.005, 0.02)	0.060 (0.071, 0.314)	0	0.295 (0.236, 0.81)	0.033 (0.031, 0.09)
	Procellariids	0.006 (0.015, 0.08)	0.001 (0.007, 0.04)	0.005 (0.021, 0.1)	0	-	-	0.197 (0.225, 0.72)	0.068 (0.061, 0.19)
	Skuas	0.020 (0.040, 0.23)	0.006 (0.018, 0.08)	0.025 (0.052, 0.24)	0	-	-	0.04 (0.069, 0.23)	0.07 (0.106, 0.550)
	Total flock	0.421 (0.259, 0.89)	0.055 (0.093, 0.08)	0.093 (0.105, 0.38)	0.008 (0.018, 0.08)	0.370 (0.290, 0.598)	0.135	0.793 (0.192, 1.0)	0.199 (0.157, 0.6)

Table A8.5 Explanatory factors of EDC variability of roundfish: parameters estimates with standard errors (S.E.) and p-values for the final model for Gannets and large gulls.

	<i>Parameter estimate (S.E.)</i>	<i>t-value</i>	<i>p-value</i>
<i>Gannets</i>			
Intercept	-3.74 (0.29)	-12.75	<0.0001
Log(gannets +1)	1.09 (0.08)	12.16	<0.0001
<i>Large gulls</i>			
Intercept	-4.79 (0.46)	-10.37	<0.0001
Log(large gulls+1)	0.23 (0.10)	2.30	<0.05
Proportion of large gulls in the flock with Gannets	3.02 (0.54)	5.49	<0.0001
Proportion of large gulls in the flock with 'other' competitors	0.69 (0.26)	2.71	<0.001

Table A8.6 Effect size (r) of differences in number of ship followers between seasons within a bird category (diagonal), between bird categories in autumn (values above diagonal) and in spring (values below diagonal). Non-significant results are indicated in bold ($\alpha = 0.05$).

	Gannets	Large gulls	Small gulls	Procellariids	Storm petrels	Skuas	Kittiwakes
Gannets	$r=0.51$	$r=0.13$	$r=0.73$	$r=0.63$	$r=0.74$	$r=0.56$	$r=0.62$
Large gulls	$r=0.58$	$r=0.10$	$r=0.74$	$r=0.66$	$r=0.75$	$r=0.60$	$r=0.66$
Small gulls	$r=0.38$	$r=0.71$	$r=0.02$	$r=0.21$	$r=0.08$	$r=0.47$	$r=0.19$
Procellariids	$r=0.12$	$r=0.63$	$r=0.26$	$r=0.06$	$r=0.26$	$r=0.25$	$r=0.01$
Storm petrels	$r=0.48$	$r=0.76$	$r=0.12$	$r=0.37$	$r=0.07$	$r=0.50$	$r=0.25$
Skuas	$r=0.36$	$r=0.72$	$r=0.05$	$r=0.23$	$r=0.18$	$r=0.37$	$r=0.24$
Kittiwakes	$r=0.54$	$r=0.78$	$r=0.22$	$r=0.43$	$r=0.12$	$r=0.27$	$r=0.30$

Table A8.7 Length ranges (min-max) of the main discarded species and discard types (bold) in the Bay of Biscay in 2010 (modified from Fauconnet *et al.*, 2011). Total length is measured for fish, carapax length for *Nephrops norvegicus* and mantle length for cephalopods. The maximum lengths were subdivided into the maximum length of >10,000 individuals and the overall maximum length in brackets.

	<i>Reported species</i>	<i>Length range (cm)</i>	<i>Reported species</i>	<i>Length range (cm)</i>
Demersal trawlers	Roundfish	10-43 (57)	Cephalopods	3-30(32)
	Mackerel	19-33 (37)	Cuttlefish	3-12(19)
	Whiting	10-22(36)	Squid	4-30(32)
	Hake	15-43(57)	Flatfish	7-21(27)
	Pouting	10-39 (39)	Sole	7-21(27)
	Monkfish	11-30 (35)		
	Horse mackerel	10-19 (36)		
<i>Nephrops</i> trawlers	Roundfish	8-26 (27)	Invertebrates	13-33 (42)
	Hake	18-23 (33)	Norway lobster*	1.3-3.3 (4.2)
	Monkfish	8-20 (57)	Flatfish	8-23 (25)
	Pouting	9-26 (27)	Sole	8-23 (25)
	Red mullet	11-(16)	Cephalopods	3-(12)
	Horse mackerel	10-15 (36)	Squid	3-(12)
	Whiting	9-(27)		
Midwater trawlers	Roundfish	8-33 (52)	Pollack	(34)
	Monkfish	8-33 (39)	Elasmobranchs	8-43 (50)
	Haddock	15-33 (40)	Cuckoo ray	8-43 (50)
	Red gurnard	13-29 (34)	Flatfish	10-34 (42)
	John dory	12-(27)	Megrim	10-34 (42)
	Horse mackerel	8-30 (40)	Cephalopods	5-(18)
	Whiting	13-30 (40)	Squid	5-(18)
	Hake	14-(43)		
	Cod	20-(52)		
Pelagic trawlers	Roundfish	7-20 (92)	Sardine	13-(24)
	Anchovy	10-17 (19)	Chub mackerel	11-(27)
	Mackerel	8-20 (45)	Sprat	8.5-13.5 (16.5)
	Horse mackerel	7-16 (43)	Blue whiting	13-(61)
	Garfish	29-(92)	Hake	18-(54)
	Seabass	44-(62)		
Gill netters	Roundfish	13-26 (93)	Invertebrates	6-(17)
	Pollack	27-(36)	Spiny spider crab	6-(17)
	Hake	18-(93)	Flatfish	16-23 (67)
	Black sea-bream	25-(32)	Sole	16-23 (67)
	Meagre	18-(67)	Sand sole	20-23 (38)
	Seabass	20-(51)	Senegalese sole	21-(33)
	Pouting	13-26 (49)	Plaice	16-(37)
	Whiting	16-(49)	Wedge sole	17-(24)
Longliners	Roundfish	22-34 (74)	Garfish	73-(74)
	Seabass	34-(35)	Pouting	20-(30)
	Whiting	22-34 (37)	Elasmobranchs	42-(47)
	Mackerel	26-(35)	Blonde ray	42-(47)

9 Beyond discard partitioning

On n'a jamais tout vu des choses de la mer. - J. Verne

9.1 The role of discards for scavenging seabirds

The positive and negative effects of discards have been investigated more thoroughly for seabirds than for any other ecosystem component (Bicknell *et al.*, 2013; Regular *et al.*, 2013). At least 143 seabird species worldwide (52% of the global taxonomic diversity) make use of discards to some extent (Oro *et al.*, 2013). Various seabird species use discards and offal as trophic resources, and some species are believed to have increased in numbers as a result of a greater availability of food via discards (Martínez-Abraín *et al.*, 2002). They forage on both live prey and fishery wastes, eventually favouring the latter when the former becomes scarce. The most exhaustive estimate of discard consumption was given for the North Sea in the 1990s. The total amount of fishery waste (without offal, defined as organic material from gutting fish) in the North Sea was estimated at 726 200 tonnes of which seabirds consumed 255 000 tonnes. About 150 000 tonnes of invertebrates were discarded, but hardly consumed by seabirds (~9000 tonnes). In total, discards potentially supported up to 5.9 million seabirds (Garthe *et al.*, 1996). However, caution is needed, given the uncertainties in mathematically combining highly variable estimates (Stratoudakis, 1999; Chapter 8), as well as the uncertainties in the discard estimates in the North Sea. In studies of the composition and fate of catch and bycatch in a *Nephrops* fishery in the Farne Deep in the North Sea, Evans *et al.* (1994) observed that 88% of the catch was made up of discards consisting of mostly unmarketable fish (34 species) and 23 invertebrate taxa, including *Nephrops*. The authors estimate that >70% of discard was consumed by seabirds near the surface. Catchpole *et al.* (2006) found that seabirds utilised an estimated 57% of the discarded material from the English *Nephrops* fishery. An example from the Mediterranean Sea illustrates that discards may fulfil four times the energetic requirements of the local population of yellow-legged gulls (*Larus cachinnans*) (Martínez-Abraín *et al.*, 2002).

The effects of discarding and a discard ban on seabirds are extensively reviewed in Bicknell *et al.* (2013). The effects are well-studied, but the authors nevertheless identified a number of poorly understood key areas: (1) the non-breeding season, (2) immature birds, (3) junk food hypothesis (Grémillet *et al.*, 2008; Wanless *et al.*, 2005), and (4) community and ecosystem interactions.

The effects of discarding for the Belgian scavenging seabird populations have been briefly touched upon in the WAKO-II project (Courstens *et al.*, 2012) and Sotillo *et al.* (2014). The WAKO-II project initiated the interaction between the Belgian fisheries' institute (ILVO) and the Research Institute for Nature and Forest (INBO) with respect to the interaction of discards and seabirds. Preliminary investigations included four questions:

1. What is the 'mean scavenger seabird community' in the Belgian Part of the North Sea (BPNS)?
2. What is the amount of discards produced by fisheries in the BPNS?
3. What type of discards and offal are consumed by seabirds?
4. What are the energetic requirements of seabirds and what is the energetic equivalent of discards?

Details on the applied methods and results can be found in the WAKO-II report (Courstens *et al.*, 2012). In summary, the authors applied two types of calculations, based on different, literature-based experiment consumption rates (a 'minimum' and 'mean' scenario). Discards in the 'minimum' scenario were providing a sufficient amount of energy to sustain the mean scavenging seabird community, except for the breeding season (April to June) when discards delivered slightly less energy to the scavenging seabirds ($10 * 10^6$ kJ). Discards provided approximately twice the energetic requirements of scavenging seabirds in the 'mean' scenario (Figure 9.1).

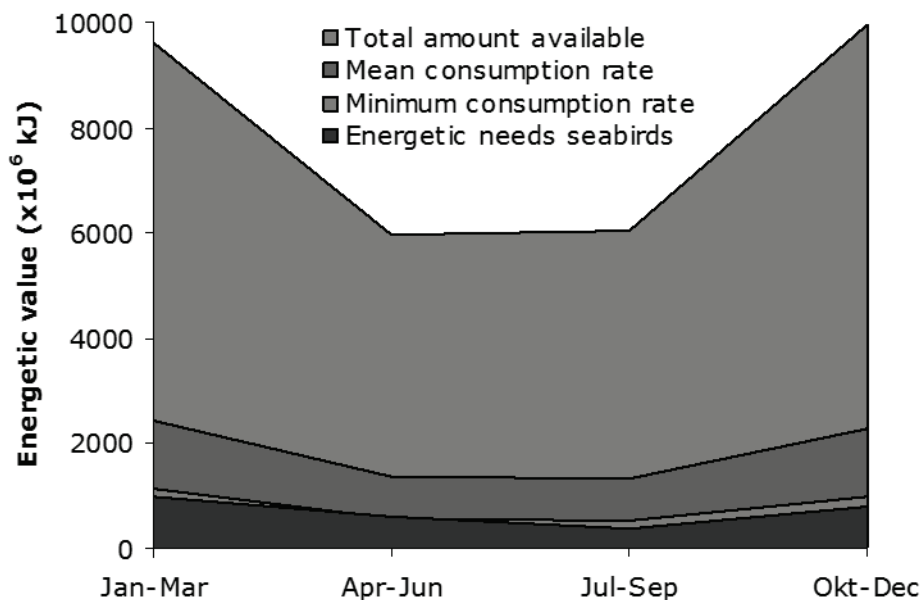


Figure 9.1 Energetics of the mean yearly amount of fisheries discards of seabirds in the Belgian Part of the North Sea (BPNS). The total energetic value of discards by quarter (Jan-Mar: January-March, Apr-Jun: April-June, Jul-Sep: July-September, Okt-Dec: October-December) varied between 6000 and 10000 $* 10^6$ kJ. Seabirds require $<1000 * 10^6$ kJ by quarter, which can mostly be fulfilled by the discards produced in the BPNS, based on two scenarios (minimum and maximum discard consumption rates). Uncertainties were high (see text).

Both Herring gull (*Larus argentatus*) and Lesser Black-backed gull (*L. fuscus*) have breeding populations in the BPNS. The mean number of breeding pairs in 2008-2011 was respectively 2534 and 4760 pairs, resulting in 14280 Lesser Black-backed gulls and 7602 Herring gulls when each pair produce one young (breeding season: April-June) or a total energetic requirement of 1650×10^6 kJ. Applying experimental discard consumption of 75% roundfish, 20% flatfish, 2% other fish, 80% offal and 15% invertebrates indicated that about two thirds of the breeding gull populations could be theoretically sustained by discards.

The estimates are preliminary and did not account for the various sources of uncertainty. The main sources of uncertainty include (1) discard estimation, (2) seabird attraction to fishing vessels and (3) experimental discard consumption rates. Discard estimation was based on a limited number of observations, but included all types of fisheries: from Dutch flatfish-directed beam trawlers to Belgian and French trammel netters. The discard estimates were calculated by applying discard rates (proportion of discards in the total catch) to the landings (Depestele *et al.*, 2012). Seabird attraction to fishing vessels was not accounted for, and experimental discard consumption were exclusively based on literature estimates. The WAKO-II study instigated EDC-investigations, resulting in single-item discard experiments (Sotillo *et al.*, 2014) and multi-item discard experiments (chapter 8).

Single-item discard experiments consisted of repeatedly discarding single items and registering the fate of the discards in relation to the flock of scavenging seabirds behind the vessel. The main contribution of the single-item experiments relate to the scavenging behaviour of breeding gull populations in Zeebrugge during the breeding season. To understand the importance of discards for local Herring and Lesser-black backed gull populations, single-item discard experiments were performed at four offshore distances from the gullery of the Port of Zeebrugge, at four different stages of a breeding season (May to August 2011). Flock composition was compared during discarding to the distribution of Herring and Lesser Black-backed gulls with respect to offshore distance from the colony as reflected in the INBO-dataset of standardized ship-based surveys (2002 - 2013). Consumption of discards depended on the type of fish that was discarded, but prey selectivity by adults was reduced during the chick rearing stage. A generalized linear mixed model identified the number of scavengers following the vessel, the proportion of adults and of Herring gulls in the flock, and the frequency of food robbery events interacting with the stage of the breeding season as affecting the variation in flatfish consumption. Shifts in scavenger flock composition and discards consumption between stages of the breeding season were likely linked to variations in food requirements of the gull population along the season, and to dispersal patterns towards the end of summer. Nutrient requirements of breeding adults peak during the chick rearing stage, making this a key period in terms of dependence of the breeding parents on discarded fish as food source.

9.2 The role of discards from benthic and other perspectives

Whether discards have played a similar role for marine scavengers in the sea as for scavenging seabirds is not well understood (Kaiser & Hiddink, 2007). Generally, the fate of discarded organisms after being submerged in the water is less clear (Chapter 8, PARTIM II, Wassenberg & Hill, 1990). This chapter sheds a light on the limited knowledge available on the fate of discards after bird scavenging and potential survival (Figure 9.2).

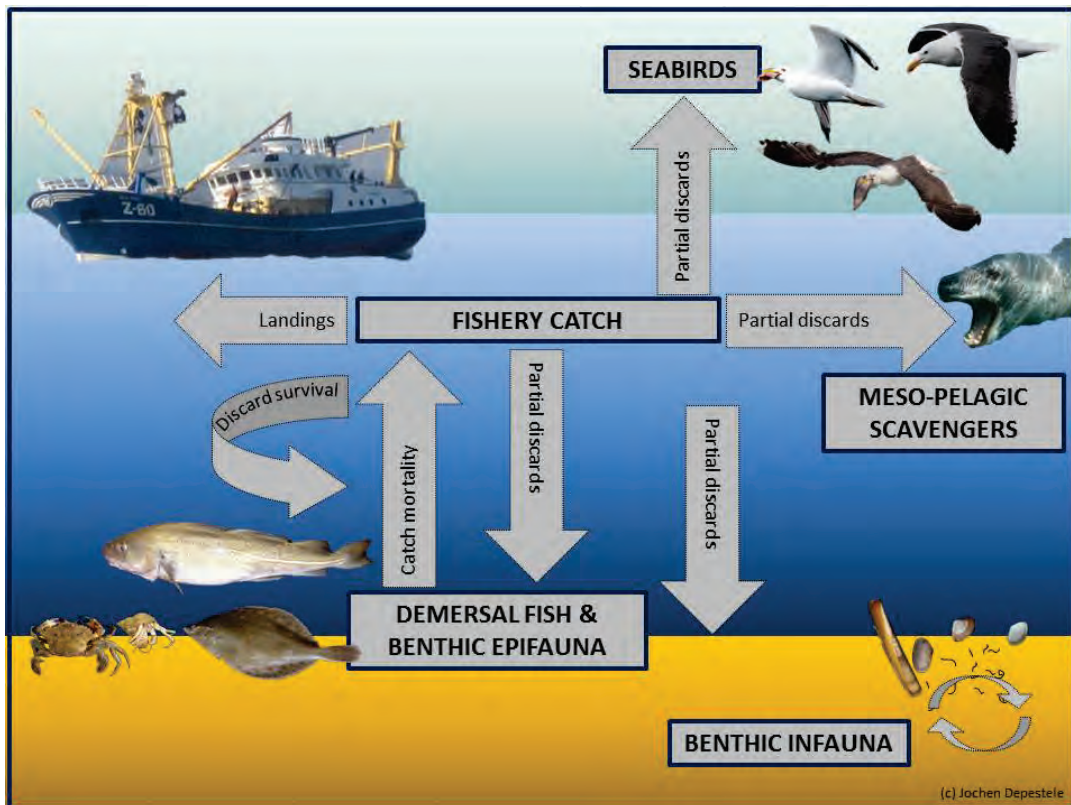


Figure 9.2 Schematic representation of the endpoints of fishery catches. Discarded organisms can be landed, or discarded. The fate of discards can be the consumption by scavenging seabirds and meso-pelagic scavengers. When they are not consumed, they reach the seafloor to the advantage of benthic scavengers or they survive the capture-and-discard process and return to the fish or invertebrate community.

9.2.1 What is a marine scavenger?

Marine scavengers are defined by Britton & Morton (1994) 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’*.

Bengston (2002) suggests that *“Scavenging and predation are often two sides of the same behavior, and detritus feeders are bound to engulf countless living microbes. Most organisms are not confined to a single mode of life, so the same organism may be predator, scavenger, parasite, etc.- and, of*

course, prey. Phenomena in nature tend to have fuzzy edges, and terminology should not lead us to forget that.” Further “fuzzy edges” exist between scavengers and filter feeders, and Walker & Bambach (1974) point out that scavenging is not a sharply defined feeding category, but merges with that of deposit feeders. Filter feeders could thus also be described as scavengers.

A facultative scavenger can be defined 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 can be defined as an animal “that relies entirely or near entirely on carrion as food resource” (Moleón *et al.*, 2014).

Britton & Morton (1994) indicate that many marine animal phyla include scavengers: Turbellaria, Nemertea, Nematoda, Polychaeta, Mollusca, Arthropoda, Echinodermata, Fish, Seabirds and marine mammals. Probably very few of these could be described as ‘obligate scavengers’. Indeed, Britton & Morton (1994) suggest that they could not exist in the marine environment due the paucity of material to forage upon, although they go on to say that ‘if there are obligate scavengers among marine animals, they will most likely be found among the Crustacea and the Gastropoda’. Their focus was on lysianassoid Amphipoda and nassariid Gastropoda.

This absence of obligate scavengers is contested by Kaiser & Moore (1999) who suggest that the lysianassoid amphipod *Orchomene nanus* is a good candidate for an ‘obligate scavenger’, and one that makes use of discarded fish. Ruxton & Houston (2004) also demonstrated theoretically that obligate scavengers could exist in marine environments. The scavenging isopod *Natatolana borealis* would be another possible candidate (Wong & Moore, 1996).

Among fish species, the most obvious candidate would be the hagfish. 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 (*Melanogrammus 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).

Finally, Beasley *et al.* (2012), suggest the seafloor environment may have encouraged the evolution of a more diverse range of facultative scavengers that make more use of carrion in their diet than occurs in terrestrial ecosystems. Following the suggestions from ICES (2015a), ‘scavengers’ are recognized to exist on a continuum between those that are close to obligate scavengers through to predators that will occasionally scavenge.

Organisms that are discarded in the sea, and which are not eaten by seabirds, may survive the catching process, or may be eaten by marine scavengers in the sea. The discards follow the reverse

pathway from being caught. They first become available to meso-pelagic scavengers in the water column, after which they reach the seabed to be a potential food source for demersal fish or epibenthic invertebrates. When these scavengers have not taken profit of the discards, they likely start to decompose and are being incorporated in the seabed to the benefit of infaunal macro- or meiobenthos, or the nutrient cycle. These consecutive steps are described below in the order that discards become available to the ecosystem.

9.2.2 *Scavengers in the water column*

Some discards may be scavenged upon on the sea surface or in the water column by marine mammals (dolphins, killer whales, etc.) or elasmobranchs (Luque *et al.*, 2006; Pon *et al.*, 2012; Svane, 2005; Wassenberg & Hill, 1990). White-sided dolphins, for instance, may be caught in midwater trawls during the night, as they scavenge upon escaping fish from trawls during the night while avoiding competition with gannets (Couperus, 1997; Morizur *et al.*, 1999). The observed scavenging or predation in the water column is predominantly related to organisms which are highly mobile and capable of following fishing vessels during their operations. The effects of discards-generated carrion may be ephemeral, but may as well be of importance to certain species populations or individuals which are specialised in capturing this easy food source. The food items are primarily fish that escape from the fishing nets before being hauled or fish that is floating at the sea surface after being discarded (Hill & Wassenberg, 2000). Fish and cephalopods may be floating at the sea surface and are known to sink more slowly. About 33 % of the fish and 45 % of the cephalopods were floating after being discarded from prawn trawlers in Australia (Hill & Wassenberg, 2000). The buoyancy of smaller fish was higher than that of larger. Roundfish with a swim bladder may also be sinking slower than flatfish (own observations). Sinking rates of crustaceans vary from 4.5 to 8.4 m per minute (Wassenberg & Hill, 1990). While seabirds considerably affect the amount of discards that sink to the seabed, it remains unclear whether meso-pelagic scavengers substantially reduce the discard availability for scavengers on the seabed. Aside from roundfish floating on the sea surface, it may be expected that most discards sink fairly rapidly to the seabed where they become inaccessible to (meso-)pelagic scavengers (Beasley *et al.*, 2012). One European study conducted experiments to evaluate midwater scavenging. Scavenging was found a variables rates, but was mainly higher at continental slope than in deeper waters (>200m). Generally low catch rates and high proportions of intact baits at the longlines indicated that scavenging in the midwater is relatively rare or at least patchily distributed.

9.2.3 Scavengers on the seabed

Key scavengers were identified from catch compositions of baited pot fisheries, as these species are also attracted to dead carrion on the seabed. Landings of pot fisheries were examined in ICES Subdivision IVa and VIIa over a ten-year period (2003-2013) based on the STECF-dataset (datacollection.jrc.ec.europa.eu). Mobile invertebrate species that were landed in >1 tonne annually were considered as important scavengers (Table 9.1).

Table 9.1 Landings of mobile (commercial) species from pot fisheries between 2003-2013, based on STECF data (datacollection.jrc.ec.europa.eu).

Scientific name	Common name	Yearly landings (tonnes)
<i>Cancer pagurus</i>	Edible crab	194.0
<i>Maja squinado</i>	Spinous spider crab	97.8
<i>Nephrops norvegicus</i>	Norway lobster	71.5
<i>Homarus gammarus</i>	European lobster	25.5
<i>Chaceon affinis</i>	Deep-sea red crab	19.2
<i>Palinurus spp</i>	Palinurid spiny lobsters nei	12.7
<i>Pandalus borealis</i>	Northern prawn	11.4
<i>Palaemon serratus</i>	Common prawn	9.1

Key scavengers were also identified 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 mainly identified on the basis of their numerical dominance and only indirectly on their dependence on discards. Studies that investigated the aggregation or increased abundances of scavengers after a trawl passage were excluded as mortality in the tow path results in different bait items than discarding. Ten studies in the Atlantic and Mediterranean Seas were summarized in Table 9.2.

Table 9.2 Discard scavenging studies in the NE Atlantic reviewed by WGECO (ICES, 2015g).

	Bergmann <i>et al.</i> , 2002	Catchpole <i>et al.</i> , 2006	Groenewold & Fonds, 2000	Jenkins <i>et al.</i> , 2004	Kaiser & Spencer, 1996	Martinez <i>et al.</i> , 2011	Moore & Howarth, 1996	Nickell & Moore, 1991	Ramsay <i>et al.</i> , 1997	Veale <i>et al.</i> , 2000	
Location	Clyde Sea	North Sea	North Sea	Irish Sea	Irish Sea	North Sea	Clyde Sea	Clyde Sea	Irish Sea	Irish Sea	
Attraction time (h)	<48	2	<48	<96	?	<7	24	24	<76	<96	
Top five scavenging species by sampling gear and based on the numbers attracted to the bait	Baited traps	<i>Asterias rubens</i> , <i>Buccinum undatum</i> and <i>Neptunea antiqua</i> , <i>Carcinus maenas</i> , <i>Nephrops norvegicus</i> , <i>Pagurus bernhardus</i>	<i>Asterias rubens</i> , <i>Buccinum undatum</i> , <i>Liocarcinus depurator</i> , <i>Myxine glutinosa</i> , <i>Pagurus bernhardus</i>	<i>Asterias rubens</i> , <i>Buccinum undatum</i> , <i>Crangon crangon</i> , <i>Liocarcinus holsatus</i> , <i>Pagurus bernhardus</i>	-	<i>Asterias rubens</i> , <i>Buccinum undatum</i> , <i>Limanda limanda</i> , <i>Merlangius merlangus</i> , <i>Pagurus bernhardus</i>	-	<i>Asterias rubens</i> , <i>Buccinum undatum</i> , <i>Carcinus arcinus maenas</i> , <i>Necora puber</i> , <i>Pagurus bernhardus</i>	<i>Buccinum undatum</i> , <i>Pagurus bernhardus</i> , <i>Natanolana borealis</i> , <i>Pandalus montagui</i> , <i>Scopelocheirus hopei</i>	<i>Buccinum undatum</i> , <i>Colus gracilis</i> , <i>Hemimysis lamornae</i> , <i>Natanolana borealis</i> , <i>Processa nouveliholt hui</i>	-
	Baited cameras or divers	<i>Brachyura</i> , <i>Cancer pagurus</i> , <i>Carcinus maenas</i> , <i>Crangonidae</i> , <i>Gobiidae</i>	-	-	<i>Callionymus lyra</i> , <i>Cancer pagurus</i> , Flatfish, <i>Majoidea</i> , <i>Ophiocomina nigra</i>		<i>Limanda limanda</i> , <i>Merlangius merlangus</i> , <i>Melanogrammus aeglefinus</i> , <i>Myxine glutinosa</i> , <i>Pleuronectes platessa</i>	-	-	<i>Buccinum undatum</i> , <i>Cancer pagurus</i> , <i>Liocarcinus spp.</i> , <i>Ophiura spp.</i> , <i>Pagurus bernhardus</i>	<i>Asterias rubens</i> , <i>Astropecten irregularis</i> , <i>Callionymus lyra</i> , <i>Liocarcinus spp.</i> , <i>Pagurus spp.</i>

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, six taxa occurred in >2 studies: *Buccinum undatum* (8), *Pagurus bernhardus* (8), *Asterias rubens* (6), *Cancer pagurus* (3), *Liocarcinus* sp. (4) and *Carcinus maenas* (3). Several factors influenced the number of identified key scavengers, such as background densities and their spatio-temporal variation.

Seasonal and diurnal feeding patterns affect food partitioning. Ramsay *et al.* (1997) for instance illustrated that *Liocarcinus* sp. increase their scavenging activity during the night, while Nickell & Moore (1991) highlight that the monthly catch of *Pandalina brevisrostris* 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 & Moore (1991), but Groenewold & Fonds (2000) showed that the consumption rates varied due to temperature differences (hence seasonally). 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 sampling gear also affects the number of epibenthic scavengers, such as the low number of *Cancer pagurus* and the under-representation of fish guilds. The entrances of the traps usually measures between 25 and 70 mm preventing larger fish to enter the traps. Groenewold and Fonds (2000) tested different types of traps, and concluded that a combination of transparent tube traps, Danish prawn traps, and small (transparent plastic) amphipod traps appeared to be most suitable to capture the suite of scavenging organisms. Several other studies also used a combination of *Nephrops* creels and funnel traps (Bergmann *et al.*, 2002; Catchpole *et al.*, 2006; Nickell & Moore, 1991). The mesh size used in these traps is the main factor determining the abundance and diversity of the retained species. Small meshed funnel traps usually retained the smaller scavengers (amphipods and isopods), while the larger meshed traps logically retained only the larger organisms (crustaceans, gastropods and occasionally fish). The catch efficiency of different trap types was an important determinant for the evaluation of a scavenger's abundance at the bait. Time-lapse camera observations complement these observations, and are less prone to catch efficiencies. They yield useful insights into the arrival times and residence time of larger, more mobile species.

The epibenthic species which were identified from the landings of commercial pot fisheries and baited experiments may therefore provide a useful indication of the most important scavengers, but should be considered with care, as they may have unintentionally under- or over-estimated the importance of species which are not easily captured. These lists nevertheless indicate which epibenthic invertebrates can be expected to take profit from discards arriving on the seabed.

The partitioning of discards among the identified demersal fish and epibenthic scavengers further depends on their numerical abundance as an indicator of their competitive advantage to others, the time of attraction to the bait (fast species may profit more) and metabolic requirements.

The importance of discards and/or carrion for demersal fish and epibenthic invertebrates is relatively unknown, but as shown above, these species' populations and the benthic community structure and functioning is potentially affected by discards. The modelling exercise of Heath *et al.* (2014a) showed that scavenging 'benthos' was affected by discards, as was concluded by Catchpole *et al.* (2006). Groenewold and Fonds (2000), in contrast, concluded that the direct importance of discards as additional food for scavengers was relatively small, albeit relatively larger for scavenging fish than for invertebrates.

9.2.4 *Scavengers in the seabed*

Scavengers that are active on or close to the seabed (demersal fish and epibenthic scavengers) arrive at the discards within a couple of days. However, by the time they have arrived, the discards are already on the seabed and available to the communities at the sediment-water interface, i.e. organisms 'in' the seabed. The effects of discards on infaunal community structures and functioning are largely unexplored, although isolated studies indicate that infaunal community responses are fast and diverse. An experimental study in the Tagus estuary shows that bacteria responded within 2h to artificially deposited discards (*Crangon crangon*) by increasing their densities (Franco *et al.*, 2008). Nematode communities responded within 6h by changing the vertical distribution of the dominant groups. The input of organic matter to communities of micro, meio to macrofauna has been examined for phytoplankton blooms in continental shelf areas (Provoost *et al.*, 2013) or for whale falls in the deep sea (Amon *et al.*, 2013), but is fairly unknown for discards. Based on inferences from these other sources of organic matter, it is clear that the respiration of carbon strongly depends on the quality and quantities of the organic matter, as well as the environmental conditions (e.g. temperature and substrate) (Franco *et al.*, 2010). The partitioning of discards among several infaunal groups (micro-, meio- and macrobenthos) can strongly differ among sites as well. The decomposition of discards at the sediment surface may lead to alteration of infaunal communities (Franco *et al.*, 2008; Lenihan & Peterson, 1998), but the potential effect is expected to be highly spatially and temporally variable, reflecting various biogeochemical changes in the sediment. Decomposition of the discards is highly localised and associated with oxygen stress underlying the dead organic material. This will not only affect the communities, but also mineralisation processes in the sediment (Aller, 1988), which in turn may provide nutrients to the water column for primary production.

Deposition of discards and subsequent decomposition of organic material may affect the benthic ecosystem in the seabed, and indirectly primary production. The effect of discard deposition on the substrate and its communities remains largely unexplored, both in terms of biotic response and ecosystem functioning, and is an important knowledge gap in determining the influence of discards on the marine ecosystem.



*Conclusions,
considerations
and future challenges*

GENERAL DISCUSSION

Discards from human, stock and ecosystem perspective

10 Reflections on the fate of discards

The voyage of discovery is not in seeking new landscapes but in having new eyes. – M. Proust

10.1 Towards the quantification of discards

10.1.1 *Conclusions and considerations*

Fishermen rarely succeed in catching exclusively fish which is suitable for human consumption and which can be legally landed. The catch often comprises species which are returned to the sea for a variety of reasons. The quantities and composition of the catch is driven by three main factors (Fauconnet, 2014). First, the exploitation pattern determines the suite of environmental variables and the suite of species that can be encountered. Exploitation patterns relate to choices of fishermen for particular locations and periods at several hierarchical scales and to the spatio-temporal organisation of fisheries management. Where and when are fishermen allowed to fish? Where do they want to go fishing? Can they deploy their fishing gear in that location at that time (seasons, time of the day, weather conditions, etc.)? The second major determinant of the catch is the selectivity of the gear. The construction and deployment of a gear relates specifically to the catching process and interacts with the behaviour and habitat of marine organisms. The deployment of a demersal trawl will result in completely different interactions with the marine environment than fishing with gill nets which relies primarily on the active behaviour of marine organisms rather than the chasing process of fishermen. The third and last factor that drives the quantities and composition of the catch is the least related to the marine environment. This factor is primarily driven by the market, namely the utilization of the catch. Catch utilization is the decision of fishermen to keep the catch or to return it to the sea. The third factor, catch utilization, is the driver for distinguishing between commercial and non-commercial species. Pouting is a typical and recurrent example of a species which is discarded because of its low market value.

These three factors results in highly variable landings and discards. Chapter 4 of this thesis focused on the interface between gear selectivity and catch utilization. The influence of exploitation pattern on discards was largely excluded by focusing on flatfish-directed beam trawl fishery in ICES Division IVc (southern North Sea). The underlying rationale aimed to disentangle gear selectivity and catch utilization as a driving factor for discarding. Catch utilization decisions were based on market and policy-driven motivators (seasonal variations in market prize and regulations on allowable landings

per trip). Four commercial fish species were selected as a case study: sole, plaice, cod and whiting. Size was a primary driver for the discards of sole and plaice in the Belgian beam trawl fishery, while the discards of cod and whiting also constituted marketable fish, primarily due to landing limitations. The discards of plaice were only driven by size, which was contrasting earlier findings in the Netherlands and the UK. The exploitation patterns as well as national regulations may have caused this difference.

The aim of disentangling the causes of discard variability was to investigate whether an increased understanding of the causes of discarding could assist in constructing models to predict discards of commercial fish species, which could, for instance, be used in addressing the MSFD-requirements in the BPNS, namely the prediction of discard levels in the BPNS as well as its ecosystem consequences. The predictive capacity of the models was, however, limited and was partially due to the limited observer coverage and hence the limited number of sampled trips to detect discarding patterns.

The ability to predict discards from fish size, market price and regulations on landings' restrictions was further complicated by the introduction of a new legislation which prohibited highgrading in 2013 (EU, 2013b). Highgrading or discarding of marketable fish implies that only the most valuable fish are retained in order to maximise the returns of the fisheries' catches. The value of marketable plaice for instance may be low during spring due to its poor condition directly after spawning. The low market value may be a causal driver to discard marketable plaice to save quota for later when the fish are thickening and market prices are higher. Similarly, marketable megrim (*Lepidorhombus whiffiagonis*) may have been discarded due to bruises in the mixed demersal fisheries in the northern North Sea. The introduction of the new legislation, however, could not prevent that marketable megrim were not discarded (MacDonald *et al.*, 2014b). Highgrading is illegal, but potentially still occurring, which in any case complicates the ability to predict discards from marketable fish. A similar issue may be prevalent when the landing obligation will come into force for demersal species in early 2016. The introduction of the new legislation may reveal for instance that discards are lower than expected from discard observer programmes. However, if enforcement and control of the new legislation is limited, it remains unknown whether lower discard levels are due to inaccurate discard estimation or illegal practices.

The limited availability of precise and accurate discard data is not solely a Belgian issue, but clearly apparent across most, if not all, European member states (chapter 2). The number of landings that were covered by discard estimates was <60% in the discard-database of STECF. The number of stocks that used discards in analytical assessments was also limited in the ICES-database, although progress was made within ICES to stimulate and improve the uptake of discard estimates in stock

assessments. It may be expected that continuing efforts will increase the use of discard data in stock assessments (ICES, 2015b), although the financial limitations of discard observer programmes may hamper uptake as was the case for discards of anglerfish in VIIab and plaice in VIIe-f (ICES, 2015c; 2015d). The statement of Holt (1895) in the late nineteenth century still seems to hold true in the twenty-first century:

'It is impossible to estimate the quantity of small fish that is destroyed since it is impossible to estimate the amount that is shovelled overboard, dead or dying.'

Catch (landings and discards) data are a cornerstone of European fisheries management, and are currently primarily based on onboard observer programmes, complemented with onshore sampling (Allard & Chouinard, 1997; Cotter & Pilling, 2007; Shelton *et al.*, 2012). The limited coverage of the fishing fleet and/or activities have, however, stimulated the development of a range of new methodologies (chapter 3; Mangi *et al.*, 2013), such as questionnaires for recreational fishing (Zarauz *et al.*, 2015) and automated sampling. The two major new developments are: (1) increasing fishermen's responsibility through self-sampling schemes or other ways of including LEK and (2) REM as a highly technical 'big brother' system. These methodologies focus on technical developments, but may be applied more effectively when they are coupled with an appropriate management system, which may be another (indirect) way of obtaining valuable information to increase fishermen's participation and/or responsibility. Investigations of the current on-board observer programmes are ongoing and parallel to the development of the new technologies.

The state-of-the-art in methodologies to estimate the total amount and composition of discards shows that research focus is primarily directed towards cost reduction and a higher fleet coverage. The new methodologies open scope for the near future, although each methodology has its strengths and its weaknesses, as summarized in Table 10.1.

The main take-home message from PARTIM I stresses that the scientific challenges in investigating fisheries' discards remain focused on methodologies to quantify discards of commercial fish species. Scientific understanding of the drivers of discarding is increasing for several European fisheries, including the Belgian beam-trawl fishery, and remains a cornerstone of European fisheries management. The increased understanding, however, does not lead directly into more accurate discard estimations. The development of technical means is therefore prompted by the scientific community, whereby fisheries management attempts to complement these with alternative regimes and legislation on discarding practices (landing obligation). These developments are crucial as discards contributed substantially to fishing mortality of several commercial stocks. They are, however, focused on a limited number of commercial species. The estimation of discarded quantities

of species with less or no commercial value hardly receives attention in European fisheries science and management. Several member states nevertheless indicated that discarded quantities are substantial for several species. Discard data on non-commercial species in the Belgian fishery were sparse and to a limited to a low number of isolated case studies. The discarded number of species and individuals were highly variable, but may account for >50% of the catch by weight and/or number.

Table 10.1 Advantages and disadvantages of the developing new methodologies of fisheries dependent data and complementing service from fisheries independent data, based on the literature overview above.

	Self-sampling	FDF and electronic monitoring	On-board observers	Modelling approaches
Type of data collected	Simplified data from target species (weight or length-measurements)	Number and/or length of single-species, can discriminate between discards and landings	Data from TAC species to all discarded species (weight, length, age, other direct biological samples)	Data from target species to all fish species sampled in surveys (TAC and non-TAC)
Cost	Low (to high if fishermen are paid)	Low-high (depending on fleet coverage)	High	Low
Fleet coverage	Potentially high, but biased on participation	Potentially high (24/7)	Low; Hampered on very small vessels	None (survey-based) to low
Validation	Depending on trust, and fishermen's involvement and responsibility in the management process	Regular weight validation required by on-board observers	Observer bias Sound statistical design required	Models require assessing its quality and reliability with 'independent' data (Augusiak <i>et al.</i> , 2014; Feeley & Silman, 2011)
Fishermen's participation	High	High, though 'big brother system'	High	Low
Detection probability of 'rare' events, e.g. porpoise by-catch in gillnets	Depending on trust between fishermen and scientists	High	Moderate	Very low
Examples	Hoare <i>et al.</i> , 2011; Kraan <i>et al.</i> , 2013; Pennington & Helle, 2011	Kindt-Larsen <i>et al.</i> , 2011; Needle <i>et al.</i> , 2014; Ulrich <i>et al.</i> , 2015; van Helmond <i>et al.</i> , 2014	Current practice; Benoit & Allard, 2009; Liggins <i>et al.</i> , 1997	Aarts & Poos, 2009; Casey, 1996; Heath & Cook, 2015; Shephard <i>et al.</i> , 2015

10.1.2 *Future challenges*

The main caveat to unravel the fate of discards is the lack of discard quantification at the level of the fishing activities on a regional as well as a local scale. Resolving the lack of precise and accurate discard data is a primary challenge for the fisheries' science community. The investigations of chapter 2 illustrated that the contribution of discards to fishing mortality generally increased from high-value to low-value species in a specific fishery, but that, in contrast, the availability of discard data followed the opposite trend and decreased from high-value to low-value species.

A single, best practice for discard observation is not likely to be found, as observations may have to be tailored to the specifics of each fishery. Discard observer programmes in pelagic fleets, for instance, have limited changes in catch profiles. They may benefit a stratified sampling design based on vessels as the catch profiles across trips do not vary substantially. Stratification by vessel may not be the most accurate option for demersal fisheries where catch profiles may change substantially across fishing trips. Another example is the use of camera-systems. Camera observations are a way forward for fisheries with a limited suite of species in their catch, but may be more difficult to apply in mixed fisheries.

The two examples above illustrated that mixed fisheries are, not surprisingly, more complex to observe, because of the rapid increase of the number of parameters to be estimated. Belgian fisheries are mixed fisheries. PARTIM I illustrated the difficulties related to the observation of discards in Belgian fisheries. There is, however, much scope to improve discard observation in Belgium.

New methodologies have not been tested and reported extensively, such as self-sampling and REM-systems. The development and testing of new methodologies, based on existing experiences from neighbouring member states (e.g. the Netherlands) in conjunction with the conventional discard observer programme may reveal interesting insights, as Belgian fisheries are limited to flatfish-directed beam trawling which are widely distributed over a broad geographical area. PARTIM I highlights that these developments are highly and urgently needed.

The objectives of these tests should be clearly stipulated in advance and balanced against the requirements at national, international, stock and ecosystem level. A few examples may illustrate how the potential focus of new research projects may contribute to increased understanding of discarded quantities in Belgian fisheries:

- Can self-sampling be useful in Belgian fisheries?

Increasing the sampling coverage through self-sampling programmes may improve the precision and accuracy of discard estimation for a limited number of species on the condition that fishermen are incentivized and knowledgeable of the investigated species. The focus should thus primarily be linked to 'important' commercial species. However, if the contribution of Belgian discards (and landings?) to regional fishing mortality of a certain stocks is limited (e.g. for most roundfish species in the North Sea), should self-sampling for this species be developed in Belgian fisheries? Self-sampling is likely of importance for the stocks of plaice in the Celtic Seas (VII_{fg}), the Eastern English Channel (VII_d) and the Irish Sea (VII_a). Self-sampling may also be of importance for species with particular vulnerabilities, e.g. discarding of juvenile fish in spawning areas or discarding of rays. If self-sampling is to be tested, a priority list of the objectives and chances to succeed should be set up.

- Can Remote Electronic Monitoring (REM) be useful in Belgian fisheries?

REM-systems may have a range of advantages which have not been fully explored. Some examples may illustrate its potential:

- o CCTVs have the potential to improve the precision and accuracy in estimating discards at fleet level. The current auxiliary variables that are required to estimate discards at fleet level are limited (landings, days at sea, etc.). The correlation between the auxiliary variable and the discards may be limited. CCTVs may extend the number of auxiliary variables and may provide variables which have a good correlation with discards. Candidates are the number and/or duration of fishing operations (hauls) and/or the catch volume.
- o CCTVs may also aid the detection of outliers, such as sampling trips in localities with very high or very low discard levels (total catch volumes), and as such improve accuracy.
- o CCTVs may also be used to evaluate whether discard estimations from discard observer programmes can be transferred from one fishery to another.

REM-systems are considerably different from self-sampling or discard observer programmes. Their use in mixed fisheries is highly complicated. Species identification, for instance, may be more difficult, and require the development of innovative sorting and image processing techniques. The technical potential of this technique is not fully explored. There are indications, however, that REM-systems improve primarily the broad picture of evaluating the fishing practices (catch volumes, duration of fishing, fine-scale spatial distribution), but

lacks the possibility to provide high-resolution details at species level (age readings, species identification, RAMP, etc.). The possibility to develop REM-systems which provide details on the broad picture, while a limited number of discard observer trips are used to sample high-resolution data such as catch composition and age-data may, however, be a potential way forward, although there is still a long way to go to make it applicable in the Belgian fishery.

- How can discard estimation be improved in the Belgian marine waters?

Discard observations in the Belgian Part of the North Sea are limited in the Belgian discard observer programme. The current Belgian discard observer programme aims to fulfil the objectives of the DCF, implying that the discard estimations cannot be evaluated locally. If precise and accurate discard estimations are locally required (e.g. for the MSFD-objectives), alternative solutions should be searched for. One solution may be the development of an additional, 'local' discard observer programme, but requires additional financial resources. A 'local' discard observer programme has been set up for trammel net fisheries in the Belgian waters in the framework of the WAKO-II project (Depestele *et al.*, 2012), but these efforts were limited to the year 2012 due to financial constraints.

Other solutions require more creativity. The Dutch discard observer programme partially covers the Belgian waters. Discard estimation in the Belgian waters may be estimated from the Dutch discard observer programme if the sampled number of trips and hauls within the BPNS meet scientific requirements of precision and accuracy. The Dutch discard data may not be representative for the Belgian fishery, given different quota allocations and management regimes, but may provide a sufficient blueprint of the discards of flatfish-directed beam trawling in the BPNS as 82% of the fishing effort in the BPNS comes from Dutch beam trawl fisheries (based on VMS-pings). The discards of shrimp-directed beam trawl fisheries in the BPNS have not been routinely sampled by any member state, although its effort is at least one third of the flatfish-directed beam trawl fisheries in the BPNS (Pecceu *et al.*, 2014).

- How can discard observer programmes be improved?

The current discard observer programmes were originally based on a range of studies in the late 1990s and early 2000s. Stratified sampling by member state using fishing trips as primary sampling was prompted as the way forward for discard estimation on the basis of several national studies. A re-evaluation may indicate how to improve these programmes by questions such as: can stratification be fine-tuned or should it be abandoned for a randomised design? Can sampling be organised by regional ecosystems instead of country?

- Can the Belgian discard observer programme be extended to species with less or no commercial value?

One example may illustrate when the composition of total catch may be sampled. If discarded quantities of whiting were not used in the assessment of whiting stock covering the North Sea (whg-47d), should the discards of whiting then be estimated or should the Belgian observer programme rather focus on estimating the total discard composition, including non-commercial species, in a few hauls or collect more age-data of other species?

- What are the implications of the Reformed CFP for the estimation of fishing mortality?

A major challenge is the introduction of the landing obligation. The new regulation may bring about a range of changes in exploitation patterns, gear selectivity and catch utilization. All of them affect fishing mortality and the food availability in marine ecosystem. The practical implementation of the landing obligation will highlight the most drastical changes and, by consequence, the research priorities that should be addressed.

10.2 Short-term survival of discards

10.2.1 *Conclusions and considerations*

This thesis focused on a case study within the Belgian beam trawl fishery to provide empirical survival estimates of fish and benthic invertebrate species and assisted in developing a methodology to estimate discard survival at fleet level. These two fundamental elements contribute to the data requirements on discard survival within the reformed Common Fisheries Policy.

The first outcome of the empirical investigations relate to the use of the short-term survival estimates, as well as the factors that explain variability in short-term discard survival. The study is the first to investigate the survival for a 4 m beam trawl with chain mats, typically used in the 'eurocutter' beam trawl fishery. The discard survival of benthic invertebrates is considerably higher than for fish. The survival was high for cod (66%) and rays (72%), intermediate for plaice (48%), but low for sole (14%) during an observation period of at least three full days (72h). All observed pouting and whiting died within 24 hours. Smaller specimens had lower chances of surviving the discards, indicating the importance of covering the length range of the discards to have a sensible estimate of survival to provide management advice. A summary of short-term survival estimates was also provided for other beam-trawl fisheries, e.g. 12 m tickler chain beam trawls. The summary table (Figure A5.1)

illustrates that survival estimates are highly variable without unequivocal indications of the causes of this variation. A pre-requisite for management advice on survival within the EU Landing obligation should first provide short-term survival estimates in a wider range of fisheries and a wider range of conditions within a fishery before statements on discard survival at fleet level can be made.

Second, the possibility to examine short-term discard survival over a wide range of fishing practices (seasons, locations, etc.) was examined. When short-term survival studies are significantly correlated to survival proxies, then the data acquisition of easy-to-collect proxies should assist in obtaining discard survival estimates that can be used at fleet level. A proxy for discard survival was developed and tested using objective criteria related to physical injuries. The study (chapter 5) indicated that physical injuries may predict the survival of plaice and rays, but was insufficient for sole and cod. The inability of the developed proxy to include internal damage was a likely explanation for the limited predictive power of the survival of cod and sole. Tests were therefore initiated to develop additional criteria which do account for the internal damage of discarded organisms. Internal damage may be measured by RAMP, the Reflex Action Mortality Predictor, but requires a careful selection of the appropriate reflexes to be representative for the impairment that is induced from the fishing process. A study was conducted that evaluated an extensive set of reflexes for sole and plaice in the flatfish-directed beam trawl fishery. Preliminary conclusions resulted in seven reflexes with potential to detect impairment in beam-trawled flatfish.

10.2.2 Future challenges

The limitations to estimate the survive rate of discards are numerous. The number of empirical estimates is low due to the complex logistics and financial demands to conduct discard survival studies that representative for the fishing activities of the entire fleet. Discard survival is nevertheless highly topical given the potential exemption of the landing obligation for species which have 'high' survival rates, as demonstrated by 'scientific evidence' (EU, 2013a). High survival was not specified, but instead, left open for interpretation. The main challenge related to discard survival is to increase our general understanding and to evaluate how important survival may be for a commercially fished stock. Does discard survival matter?

The way forward is, by all means, to collect an increasing number of empirical data using a standardized and harmonically coordinated methodology in order to increase the comparability of the estimates across fisheries, species, locations and seasons. The integrated approach as formulated within ICES (2014c) clearly contributes to this end.

The collation of short-term discard survival estimates may be most easily performed and should be coupled to survival proxies in order to evaluate survival at fleet level in a scientific rigorous way and to be able to address future claims on increased discard survival by technical developments. Long-term estimates are less likely to deliver direct management advice within the proposed time frame of the implementation of the landing obligation (2015-2019), but are nevertheless required if the contribution of discarded organisms to the managed stocks are to be understood. General trends on factors that affect the long-term viability may prioritize research efforts. Scavenging seabirds, for instance, prefer to eat roundfish rather than flatfish. The long-term survival of flatfish may be less compromised in the long term than the survival of gadoids by scavenging seabirds.

The collection of empirical short-term discard survival estimates is ongoing throughout Europe for a selected number of species, e.g. plaice in the Belgian and Dutch beam trawl fishery. The selection of species and fisheries has likely resulted in a list of species which have the highest discard rates in the most important (valuable?) fisheries and which may be the choke species. Choke species in a mixed fishery are species which *choke* or compromise the opportunity to catch the quotas of other species in that fishery because of low quota.

While these studies will not be able to detect all influencing factors for all European fisheries, they will be able to indicate the major ones for the major species and fisheries, potentially length and haul duration for flatfish discards in trawl fisheries. A meta-analysis of the major trends may indicate which species and fishery are most susceptible to high mortalities from discarding and which species may survive more easily. An analysis that prioritizes which species and fisheries lead to the lowest discard survival can assist to identify for which species and fisheries are the best candidates for investigating long-term survival.

Besides a prioritization for investigating the survival of the most 'important' discard species (high discard quantities and discard low survival), there may also be the possibility to estimate the contribution of discard survival to the biomass of the stock. It may however be stressed that the need to collect discard survival data should go along with the necessity of collecting accurate and precise estimates of the discard quantities, which are up to this date only limitedly available.

If discard survival actually matters, i.e. if the survival of discarded organisms for a species-specific population contributes significantly and substantially to the long-term sustainable exploitation of a population, then the systematic collection of survival proxies may be questioned. The relevance of discard survival for a stock should, however, be a secondary priority. The first priority remains the improvement of methodologies to quantify discards despite the focus of reformed CFP on *high* survival rates.

The main take-home message on research prioritization from PARTIM II may be formulated as follows for Belgian fisheries:

- Increase the number of discard survival studies using the short-term discard survival approach focusing on the plaice as a potential choke species.
- Develop the RAMP-methodology and link it to short-term discard survival of plaice. The RAMP-survival curve should be applicable in a wide range of fishing conditions.
- Collect RAMP-scores for a specific stock using discard observer programmes.
- Combine the discard quantities of plaice with the discard survival estimates for a particular stock and quantify the contribution of discard survival to the biomass of stock. If discarded quantities are high, e.g. for plaice in the Celtic Seas (*ple-celt*), what are the implications of discard survival for the biomass of the stock?
- If discarded quantities are high and discard survival is low, are there technical measures that may improve discard survival in order to increase the contribution of discards to stock biomass?

10.3 Can we reduce discarding?

10.3.1 *Conclusions and considerations*

The vast range of discard drivers across fisheries, areas, species, national and international regulations and markets makes it difficult to develop a one-size-suits-all measure. In contrast, discard mitigation measures likely need to be tailored to each particular case. The listed discard mitigation measures were not exhaustive, but focused in the first place on the discard mitigation options that were investigated in the Belgian fisheries, mainly (if not exclusively) gear measures. Other mitigation measures, such as spatio-temporal measures, were solely mentioned to indicate that a little more thinking-out-of-the-box should be initiated to explore other, and potentially more viable discard mitigation possibilities which avoid uncertainties related to the efficacy of gear measures in various conditions as well as uncertainties related to unaccounted mortalities, such as those from avoiding gear, escaping through square mesh panels, etc. The capacity to reduce discarding may hold the greatest potential when several discard mitigation measures will be combined, and its efficacy is to persist when their introduction is properly enforced or supported within the fishermen's community through, for instance, increased responsibilities in the fisheries management.

Fishing gear was proposed as one of the simplest changes to a fishery, i.e. a measure that does require much interfering in regular fishing practices. Gear measures range from gear modifications

altering net selectivity, over modifications to the increase catch efficiencies, to replacements of the fishing gear by an alternative gear catching the same target species. Interference in fishing strategy increases with increasing change to the gear, culminating in its replacement by another gear.

In Belgian flatfish-targeting beam trawl fisheries, experimental gear trials indicated that minor changes to the gear, i.e. changes to net selectivity, may decrease the discards of roundfish and benthic invertebrates without any or with a limited loss of commercial catches. Reductions in flatfish discards could only be achieved when a loss of the target species is accepted, or when discard reductions are limited. While a vast suite of gear modifications were tested in experimental conditions, only two gear measures have actually been successfully implemented in commercial practice. Belgian beam trawlers in the North Sea did not convert to pulse trawling at the expense of their profitability, and the introduction of a limited number of trammel netters failed. Gear measures were only implemented when economic incentives were at stake (high fuel prices or stringent quota restrictions).

The discards in Belgian beam trawl fisheries remain high (chapter 2, 4); still the uptake of technical measures remains limited. The identification of possible mechanisms to change fishing practices to reduce discards is proposed as a priority for the nearby future of Belgian fishing gear technology research. Another priority research area is the follow-up of the implementation of the gear measures. Are they effective in reducing discards? The evaluation of changes in catch composition and subsequent discard levels may also require evaluation of changes in survival potential, which was not considered in the evaluation of gear measures.

Lastly, gear was proposed as a simple way of changing the catch composition and reducing discards. The potential of gear measures should be put into perspective of other variables such as the exploitation pattern and catch utilization. The development of gear measures is generally initiated from particular local conditions. Its efficacy at fleet level may differ due to differences in fishing practice between individual fishermen, spatial and temporal variation of biological, environmental and technical conditions. Appropriate follow-up schemes are indispensable to evaluate the potential of gear measures in discard management. The evaluation of the implementation of gear measures in other fisheries showed mixed results. The limited efficacy in other fisheries as well as the limited potential from gear trials indicate that gear measures likely need to be complemented with other measures if discard level in Belgian fisheries need to be reduced to 'acceptable' levels.

10.3.2 *Future challenges*

Short-term reductions of discard levels may be expected from selected experimental trials in Belgian research programmes. Its potential to effectively reduce discards at fleet level may not come about, as was the case in many fisheries throughout Europe, because of low compliance, circumvention or unintended impacts (Rochet *et al.*, 2014b; Tsagarakis *et al.*, 2013). The efficacy of technical measures to the reduction of discards was questioned (Heath & Cook, 2015), which implied the need for other mitigation measures to complement or replace them, such as real-time spatial catch management.

There is however only one way to find out in Belgian fisheries and that is uptake of one or more gear measures at fleet level. Uptake requires increased initiatives from the Belgian fishing industry and/or fisheries management. The last decades clearly indicated that implementation of gear measures in the Belgian fishing fleet was present when direct profit (fuel reduction or quatum) was at stake. A clear recommendation is to invest in the uptake of gear measures in the Belgian beam trawler fleet and valorise fishing gear technology research. If gear measures are not tested at fleet level by the fishing industry, experimental fishing gear trials are to be questioned.

Another recommendation is to test the the efficacy of the gear modifications that were introduced at fleet level. There were two gear modifications introduced, but do they work? The efficacy of the discard reduction is not warranted at fleet level, and depends on an increasingly number of variables that were not tested in research trials (e.g. stormy weather, compliance, fisherman-specific gear and behaviour, etc.). The effective implementation of net modifications and their provisioned effects at fleet level require to be demonstrated (by monitoring for instance) prior to them being accepted as the solution to the reduction of high discards levels in Belgian beam trawl fisheries. Testing the fleet-level effect of the introduced measures (large meshes in the top panel and a large mesh extension in the trawl net) is recommended.

Gear measures still interact with organisms. It may be worthwhile to consider changes in exploitation patterns to reduce discard levels and comply with the reformed CFP through spatial, real-time management to match fishing opportunities with catches. The support of fishing industry is also in this case crucial. There is little sense in continuing development if the initiatives are not put into practice. An investigation of drivers of change is therefore recommended.

10.4 Towards the quantification of the fate of discards

10.4.1 *Conclusions*

Seabirds tend to extract the most energetically rich organisms from the discards (larger fish with higher energetic content, e.g. pelagic fish such as mackerel and herring). The share of the discards which are not consumed by seabirds gradually becomes available through organisms in the sea through the sinking process. About one quarter of the discards was consumed by seabirds in the Bay of Biscay case study, although their consumption varied from negligible to substantial across foraging guilds, discard types, semesters and locations. The sinking discards had limited potential to subsidize marine scavengers on the seabed on a large scale, but food may be substantial for certain scavengers on a local scale.

The sequence by which discards are presented to other scavengers (sea surface, water column, sea bed) also determines the type of scavengers that may take profit from them: from highly mobile pelagic towards less mobile, sedentary organisms in the seabed. The sinking rate of discards is particularly high, implying that discards are scavenged at the sea surface by organisms or individuals that specialize in this type of food source, e.g. killer whales or white-sided dolphins. Discard do not reside for long in the water column, implying that scavenging in the water column is likely to be an ephemeral event rather than a substantial link in the food web.

Demersal fish and epibenthic invertebrate scavengers may consume discards within days. These species generally have an opportunistic life modus with high turn-over rate, which potentially gives them a competitive advantage from discards in comparison to other taxa of the epibenthic communities. The influence of discarding to those populations is not fully understood. A non-exhaustive literature search indicated that studies have demonstrated that these taxa scavenge upon discards, but not the importance of discards in their diet. Similarly, the effect of discards on infaunal communities may be substantial, but is largely unknown. Organic matter affects the organisms on and in the seabed, but scientific understanding of the magnitude of this influence is a key data gap in our understanding of the fate of discards.

10.4.2 *Considerations and future challenges*

10.4.2.1 *Improving the framework on discard partitioning: addressing uncertainty and variability*

The amount of discards that is extracted from the marine environment by seabirds has been quantified in a limited number of case studies. Chapter 8 partially resolves this knowledge gap by the development of a framework to evaluate the spatial and temporal aspect of this extraction. While the inclusion of spatial and temporal variability provides advances to previous methods (Catchpole *et al.*, 2006; Furness *et al.*, 2007; Kaiser & Hiddink, 2007), the uncertainty and variability remains high in several steps. The attraction of seabirds to fishing vessels is a major source of variability, which may be resolved by the recent developments in high-resolution spatial and temporal mapping of both seabird foraging patterns as well as fisheries distribution. The variability in experimental discard consumption can only be further resolved by boat-based studies, or potentially be complemented with information from bird-borne cameras. The variability was described in detail in Appendix 11.3 and highlighted that metier-specific estimates may resolve much of the variability in seabird attraction and discard consumption.

10.4.2.2 *Advancing the framework on discard partitioning: biomass and energy*

Chapter 8 predominantly focused on partitioning the number of discards, whereas ecosystem interactions are predominantly studied in terms of biomass or energy, and not in numbers. Biomass estimates could not be calculated, because of the experimental design of discard consumption experiments. These experiments typically estimate the number of food items that are swallowed by seabirds. Recent advances in diet studies (e.g. stable isotope analyses) were directed towards discards *sensu lato* and could be linked to discard types whereas discard consumption experiments can.

Length-dependent discard consumption has been demonstrated in various experiments, but were not included because it takes length-frequency data of the discards at fleet level to convert the numbers of consumed discards to a biomass of consumed discards. Biomass estimates of discards are widely available, but LFDs are not (Chapter 8). The proportion of consumed discards (EDC in numbers) cannot be used as a proportion of consumed biomass unless discard consumption is independent from discarded size (which it is not), or unless the LFDs of the discarded samples are representative for the fleet discards. If length-frequency distributions of the discarded samples are

representative for the LFDs of the discards at fleet, then it may be assumed that the length-dependency of the EDC-estimates was accounted for in estimating EDC.

Current biomass estimates of discard consumption at fleet level on a regional scale were based on this assumption (Chapter 8; Furness *et al.*, 2007; Garthe *et al.*, 1996; Kaiser *et al.*, 2007) and may introduce bias due to the size-preferences of seabirds and the size differences of discards across metiers.

Prioritizing metier-specific investigations may partially overcome this issue. The composition of discard types (roundfish, flatfish, etc.) in the Bay of Biscay, for instance, varied considerably across metiers (Figure 8.4, Figure 11.10). The contribution of gill netters to the overall discards was higher when expressed as biomass instead of numbers, as gill netters typically discard larger organisms (Morandeau *et al.*, 2014; Table A8.7). A metier-specific approach can, however, not solve all of the size-related issues as the contributions to the discards may also differ between numbers and biomass within a metier. The number of discarded roundfish by pelagic trawlers in the Bay of Biscay, for instance, was at least three times higher in the second semester than in the first (Figure 11.10). In contrast, the discarded biomasses were nearly equal in both semesters. Pelagic discards in the second semester were dominated by small species (e.g. *Sarda pilchardus*, *Sprattus sprattus*), whereas discards in the first semester also comprised larger individual such as *Trachurus trachurus*. Metier-based estimates may partially overcome the size-based bias, but not entirely.

In addition, it may be noted that seabirds extract especially the most energetically rich discards, i.e. roundfish and in particular 'shiny' (pelagic) fish species (own observations; Stienen, pers. comm.). Gannets (*Morus bassana*), for instance, are plunge diving for fish, an activity which requires an energetic investment which needs to be paid off by extracting high-energy food items from the sea. Gannets focus primarily on larger discards such as Atlantic horse mackerel (*Trachurus trachurus*, hereafter horse mackerel) rather than the smaller blue whiting (*Micromesistius poutassou*). Discards are especially in winter an additional and welcomed food source for Gannets (Moseley *et al.*, 2012; Pichegru *et al.*, 2007). The extraction of energy by seabirds may be proportionally higher than the extraction of biomass. Discards from demersal fisheries are otherwise not (or only limitedly) available to scavenging seabirds. The biomass of demersal fish may thus constitute an additional energy loss from the sea to the advantage of scavenging seabird populations (Bond & Diamond, 2010; Farmer & Leonard, 2011; Navarro *et al.*, 2009).

In conclusion, innovative solutions are thus required to evaluate discard partitioning in terms of biomass and energy rather than in numbers.

10.4.2.3 Applying the framework on discard partitioning to Belgian fisheries and other regions

The modelling framework on discard partitioning was applied to the French fisheries in the Bay of Biscay, because of the high diversity in benthic and pelagic fish assemblages, fishing fleets as well as scavenging seabirds. The case study was especially because of its good documentation, i.e. data availability of boat-based and aerial seabird census and discard monitoring of all returned organisms, including non-commercial species. Application to other regions is, in principle, possible, but requires that both discards and seabirds are well documented.

The framework was not applied to Belgian fisheries, because the discards are only limitedly monitored for species which do not fall under the DCF-requirements (PARTIM I). These discards may account for more than half of the discarded numbers or biomass, and may not be ignored. Testing the framework to other fisheries and regional seas may, however, reveal interesting insights in the potential contribution of discards to the diet of scavengers, because discarding differs considerably by regional seas. The southern North Sea is a particularly interesting case study, as the discards in this region are not dominated by seabirds' preferred food items (roundfish), but by flatfish (Borges *et al.*, 2014), because of the dominance of beam trawl fisheries in this region (Figure 10.1).

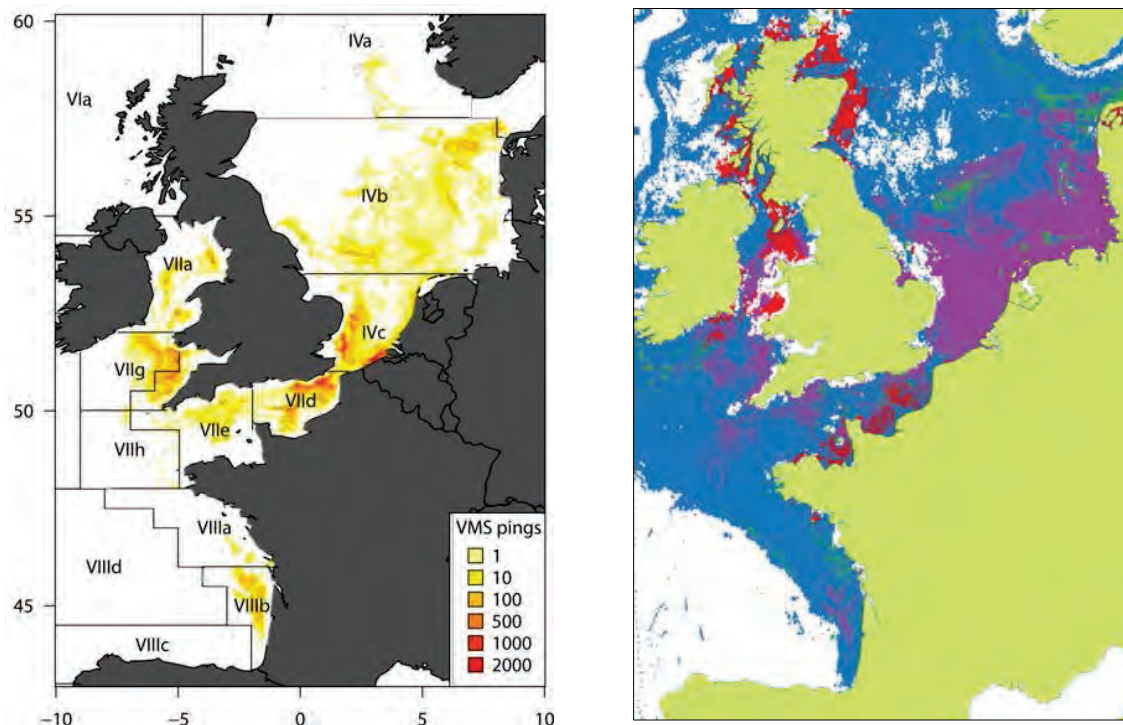


Figure 10.1 Spatial distribution of fishing effort (VMS-pings) of the Belgian fishing fleet in 2009-2013 (Vanellander *et al.*, 2015) (left) and fishing hours of several gears groups of 17 European in 2009-2012 (right) (ICES, 2014e). Countries included Belgium, Denmark, France, Germany, Ireland, the Netherlands, etc. Colour codes of the right panel corresponded to fishing effort. If 75% of the effort within a rectangle (0.05*0.05 degree) was bottom otter trawl and 25% beam trawls, then 75% of the square was blue and 25%

was red (red: dredges, green: demersal seines, purple: beam trawls, blue: bottom otter trawls).

An application of the framework developed in Chapter 8 to the southern North Sea holds much potential on the condition of discard data availability, requiring discard data from both the Belgian and the Dutch beam trawler fleets (Chapter 10.1). Data on the distribution of scavenging seabirds and the number of ship followers may originate from the European Seabirds at Sea Database (Stone *et al.*, 1995) and recent developments to track bird movements (Appendix 11.3). Experimental discard consumption has recently been investigated in the southern North Sea on-board the RV 'Zeeleeuw' and 'Belgica' (Sotillo, 2012; Sotillo *et al.*, 2014; Chapter 8).

The assessment of discarding effects in the southern North Sea may also be relevant for the evaluation of the MSFD in Belgian marine waters, as discards may affect several MSFD-descriptors, in particular Descriptor 1, 4 and 6 (biodiversity, food webs and seafloor integrity). The prerequisite for assessing the implication of discards in the BPNS, however, requires that discard data availability is further investigated (PARTIM IV).

10.4.2.4 Addressing benthic and other perspectives

The contribution of discards to marine scavengers in the sea remains a largely unexplored area. Plausible research questions to address this topic relate to (1) the identification of the key scavengers on and in the seabed, (2) quantification of the amount and composition of discards that become available to them (giving their mobility and the places where discards land after sinking), (3) quantification of the contribution of discards to the diet of the scavenging populations in terms of biomass or energy and (4) estimations of the potential community shifts due to increased scavenging populations. The potential contribution of discards to the benthic scavengers may be validated by linking the distribution of epi- and infaunal benthic communities with the fine-scale spatial distribution of fisheries as a proxy, as discards are just one of the ecosystem effects of fishing.

10.5 General reflections

This thesis addressed a topic which has long been disregarded in fisheries management: the fate of discards. The reformed Common Fisheries Policy is in the process of banning the discards of several commercial species (EU, 2013a) to reduce the unwanted mortalities and rebuild commercially exploited populations. The contribution of discards to the fishing mortality of several stocks was shown and highlighted that discard estimates cannot longer be ignored. An evaluation of the quantification of discards highlighted that estimation methods lack precision and/or accuracy, and that the availability of discard data decreased from high-value to low-value species. In contrast, the contribution of discards to fishing mortality follows the opposite trend and generally increases from high-value to low-value species. There is an urgent need to monitor the discards of all animals that are returned to the sea as their returned numbers and/or biomasses are not minor.

The landing obligation initiated the discussion that the extraction of biomass may not be favourable for commercial stocks, as a proportion of the discards may survive the capture and release process. The survival rate of discards was (and is) largely unknown for most species and fisheries. Discard survival was examined for several species of fish and benthic invertebrates in a particular case study reflecting commercial discard practices of *'eurocutter'* beam trawlers. Discarding may partially contribute to the fishing mortality of a commercial stock, but discards may, in part, also have a potential to survive the fishing process.

Science is at the brink of understanding the influence of the capture-and-release process for organisms. The collected, empirical data provides new insights in the relevance of several biological and technical factors (e.g. small fish die faster, cod may survive better than originally expected), but was nevertheless insufficient to enable extrapolation of the results to the fleet level. Discard survival experiments are complex, both logistically and financially, and require that sampling is representative to the fishing conditions of the fleet. A proxy, that significantly relates to discard survival, may resolve this issue. Physical injuries are easily measured and quantified objectively. They have potential as a proxy for the survival of benthic invertebrates, plaice and rays, but lack predictive power for other taxa. The capacity to evaluate survival may be due to the internal damages of the endured stress from the fishing process. Reflexes to external stimuli are innate responses that reflect an organism's inner damage. Several reflexes were proposed to this end. The development of a proxy based on physical injuries and reflexes holds great potential for the collection of information on discard survival at fleet level and may contribute in the move towards a better understanding of fishing effects and notably discards on the populations of commercial species.

The survival was estimated from experiments in onboard holding facilities. Long-term survival may be hampered by diseases and predation, implying that the discarded species does not contribute to the commercially exploited stock, but instead serves as food for taxa of other ecosystem components. Scavenging seabirds take a large share of the discards, about one quarter, but may vary across space, seasons, discard types (roundfish, flatfish, etc.) and foraging guilds of seabirds. The share that was not scavenged by seabirds disappeared under the water surface, and may have, in part, survived or served as food to meso-pelagic scavengers or scavengers in and on the seabed. There is still a long way to go to quantify the fate of discards after they have disappeared under the water surface.

The need to understand how discards may cause changes in single-species population and trophic interactions between species and trophic levels remains high, as discarding has increased in importance in recent years (Heath & Cooke, 2015), but it also prompts questions on the development of effective discard mitigation measures. Research on discard mitigation measures in Belgian fisheries was mostly directed towards gear-related fixes, but their implementation in the fishing fleet remained largely absent. Research efforts in the near future should therefore focus on either the implementation of the gear measures in the Belgian fishing fleet, or the development of other measures which may be more easily accepted by the fishing industry. However, as the complete elimination of discards from fishing is likely unrealistic, a better understanding of the fate of the discards remains a key priority.

The advances in this thesis have, however, shown the possibilities that are currently evolving to set up an overall framework to determine the fate of discards, and how they can be included in food chains and/or food webs. Discards may induce cascading effects in the food web and affect particular species or species groups. The effects of discards on these groups, such as scavenging commercial species (*Nephrops norvegicus*), can up to this day not be fully understood at a resolution which is sufficiently high to inform sound management decisions, but science is well on its way and incrementally increasing along the lines of Figure 1.1.

The interest of fisheries management (CFP) in the fate of discards remains focused on the human and stock perspective. Discards may, however, also substantially affect the MSFD-descriptors and play a particular role for scavenging populations of the marine ecosystem, which may in turn compromise the long-term survival of the discards. While the implementation of the ecosystem approach to marine management has recognized the ecosystem perspective (MSFD), fisheries management remains nearly exclusively focused on the human and stock perspective. Not looking at the role of discards in the marine ecosystem prevents seeing its (potential) relevance, not only for the ecosystem but also for the commercial populations inhabiting it.

This thesis compiled and advanced on several elements that are required to build an understanding of the role of discards for humans, commercial 'single-species' stocks and the marine ecosystem at large. The outlook to meet the overall aim of this thesis, i.e. the quantification of the fate of discards, can be formulated by three generic recommendations:

1. Develop discard observation programmes that meet the criteria of precision and accuracy required to estimate discards at fleet level, and complement the estimates with generic discard survival estimates at fleet level in order to simulate the contribution of discard mortality to single-species stocks (human and stock perspective).
2. Account for the discards of all animals in the newly developed observer programmes (the intersection between human and stock perspective and the ecosystem perspective).
3. Elaborate the framework of discard partitioning for ecosystem components in the water column and at the seabed in order to simulate the contribution of discards to the food requirement of several ecosystem components other than birds (ecosystem perspective).
4. Investigate the role of discards in the food web. Can discards dampen the oscillations between consumers and their resources (ecosystem perspective)?
5. Estimate the contribution of discards among other sources of mortality and food, such as mortality and changes in geochemistry in the tow path of the trawl (ecosystem perspective).

11 Appendices

11.1 Fisheries' catches (landings and discards) in stock assessments

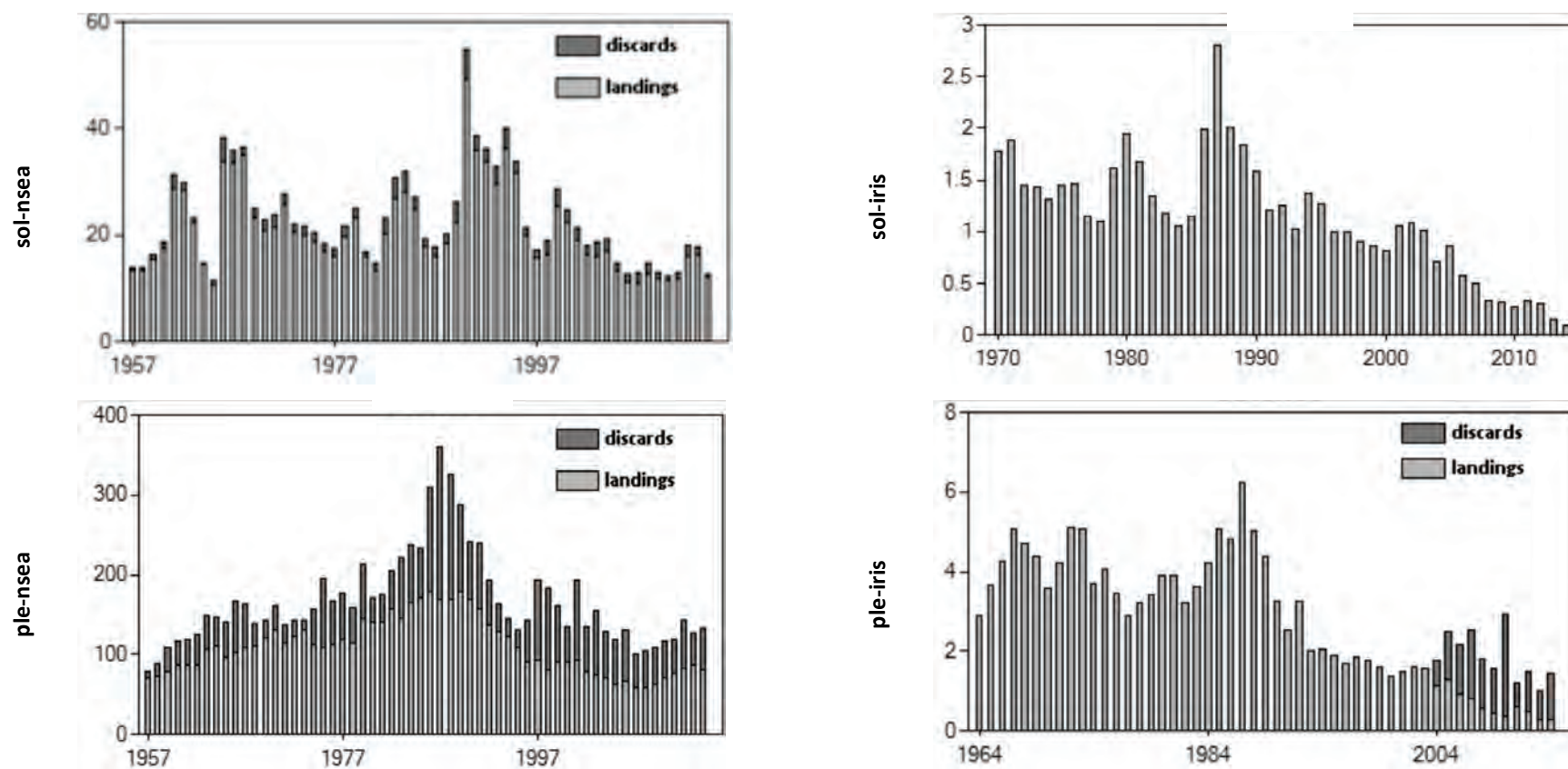


Figure 11.1 The evolution of fisheries' catches (landings and/or discards) over time and the inclusion of discards in stock assessments (ICES, 2015a). Fish stocks are described in detail in chapter 2: sol-nsea: sole in the North Sea, ple-nsea: plaice in the North Sea, sol-iris: sole in the Irish Sea, ple-iris: plaice in the Irish Sea

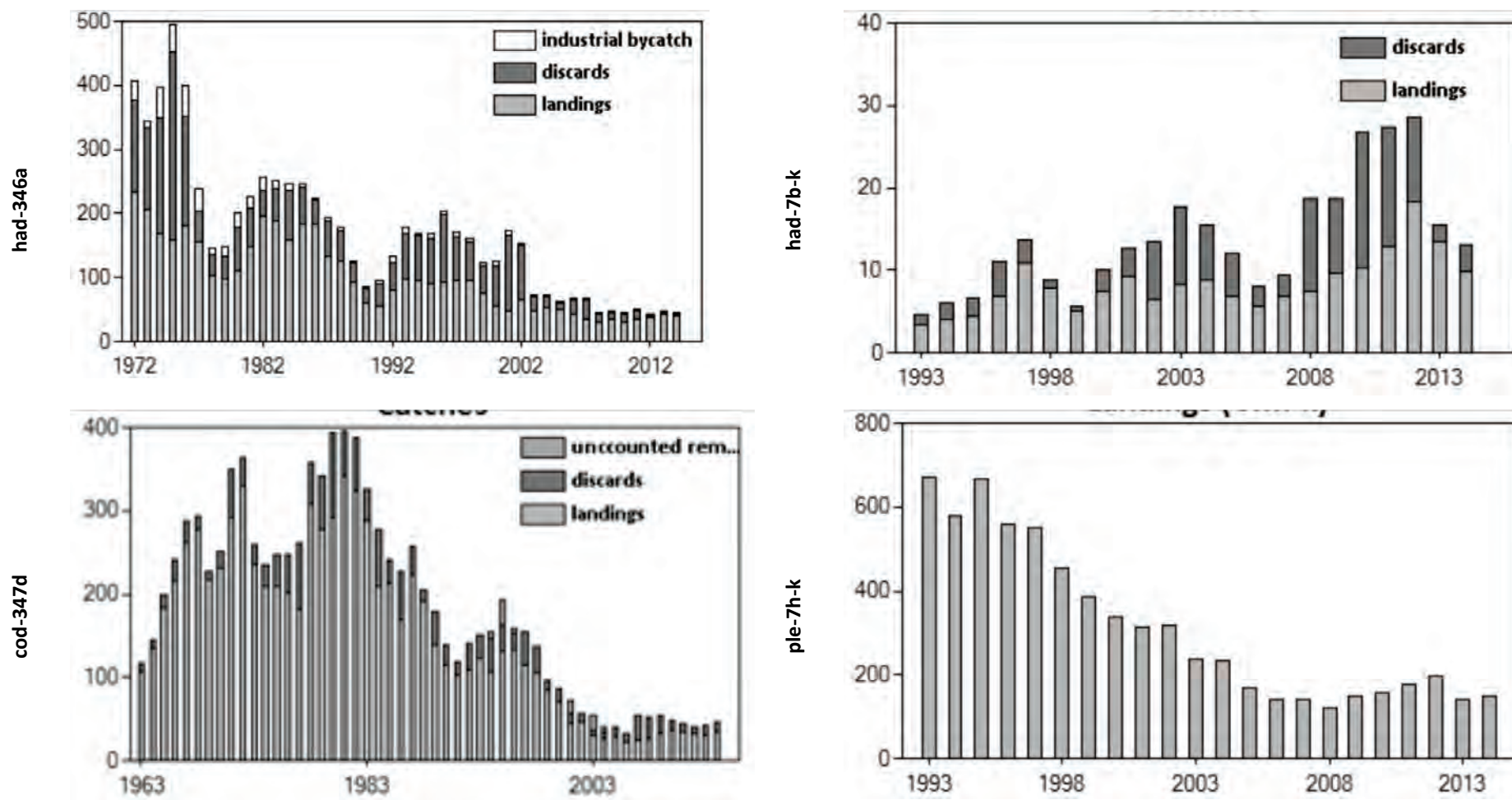


Figure 11.1 (continued). The evolution of fisheries' catches (landings and/or discards) over time and the inclusion of discards in stock assessments (ICES, 2015a). Fish stocks are described in detail in chapter 2: .had-346a: haddock in the North Sea, West of Scotland, Skagerrak; cod-347d: cod in the North Sea, Eastern English Channel, Skagerrak; had 7b-k: haddock in the Southern Celtic Seas and English Channel; ple 7h-k: plaice in the Celtic Sea South and Southwest of Ireland.

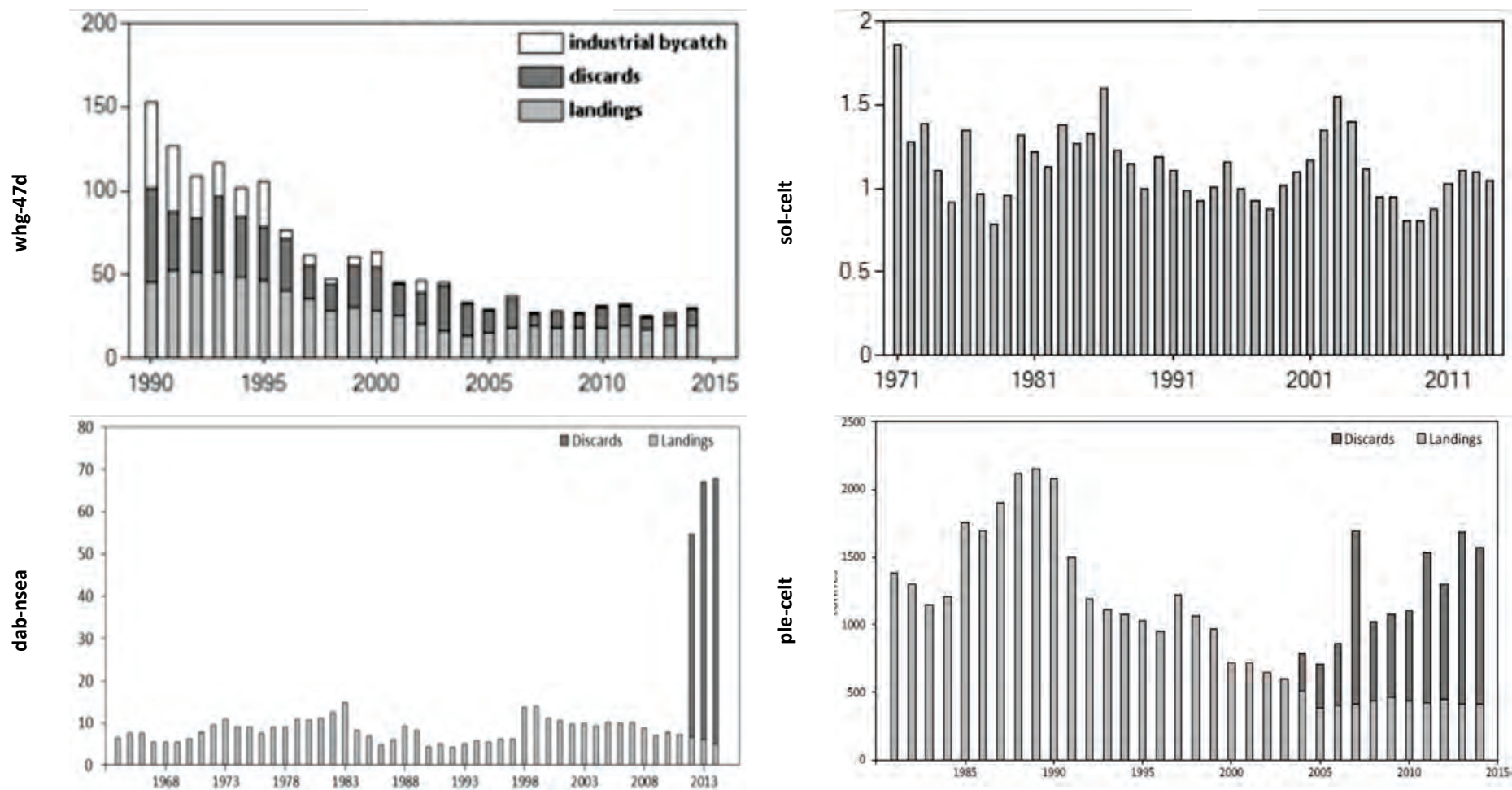


Figure 11.1 (continued). The evolution of fisheries' catches (landings and/or discards) over time and the inclusion of discards in stock assessments (ICES, 2015a). Fish stocks are described in detail in chapter 2: whg-47d: whiting in the North Sea and Eastern English Channel, dab-nsea: dab in the North Sea, sol-celt: sole in the Celtic Sea and the Bristol Channel, ple-celt: plaice in the Celtic Sea and the Bristol Channel

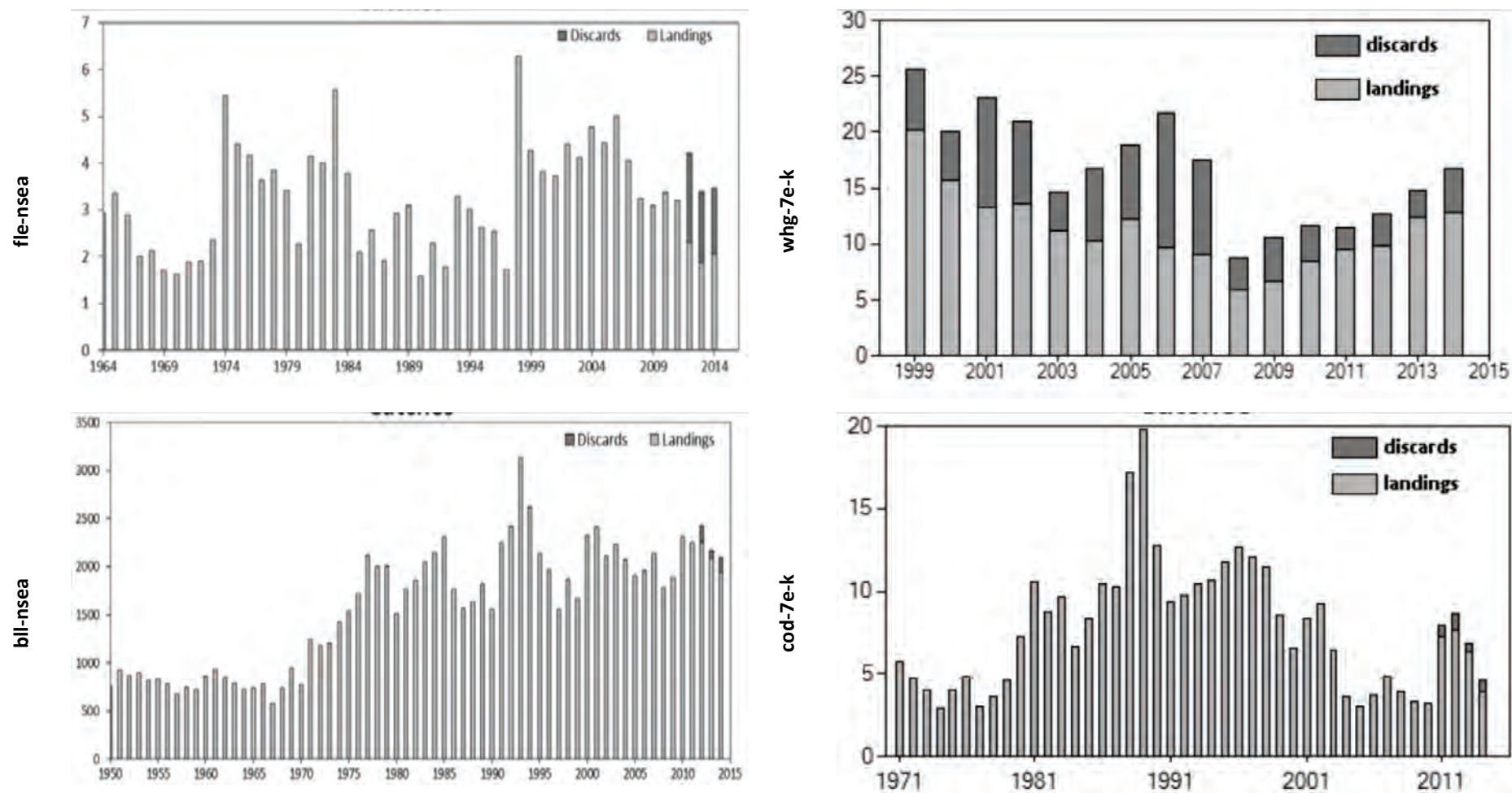


Figure 11.1 (continued). The evolution of fisheries' catches (landings and/or discards) over time and the inclusion of discards in stock assessments (ICES, 2015a). Fish stocks are described in detail in chapter 2: .fle-nsea: flounder in the North Sea, bll-nsea: brill in the North Sea, whg 7e-k: whiting in the Southern Celtic seas and Eastern English Channel, cod 7e-k: cod in the Southern Celtic seas and Eastern English Channel.

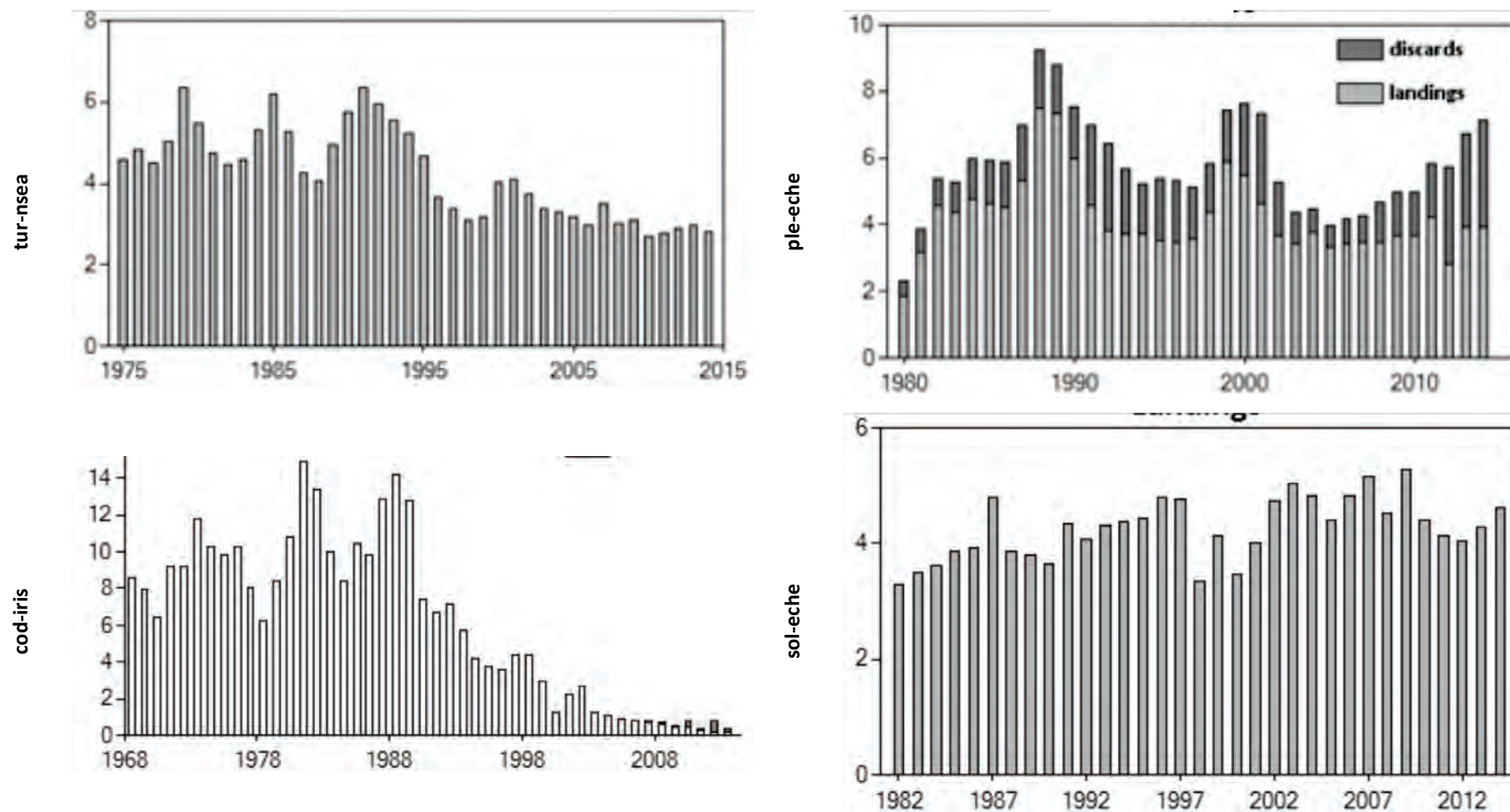


Figure 11.1 (continued). The evolution of fisheries' catches (landings and/or discards) over time and the inclusion of discards in stock assessments (ICES, 2015a). Fish stocks are described in detail in chapter 2: .tur-nsea: turbot in the North Sea, cod-iris: cod in the Irish Sea, ple-eche: plaice in the Eastern English Channel, sol-eche: sole in the Eastern English Channel.

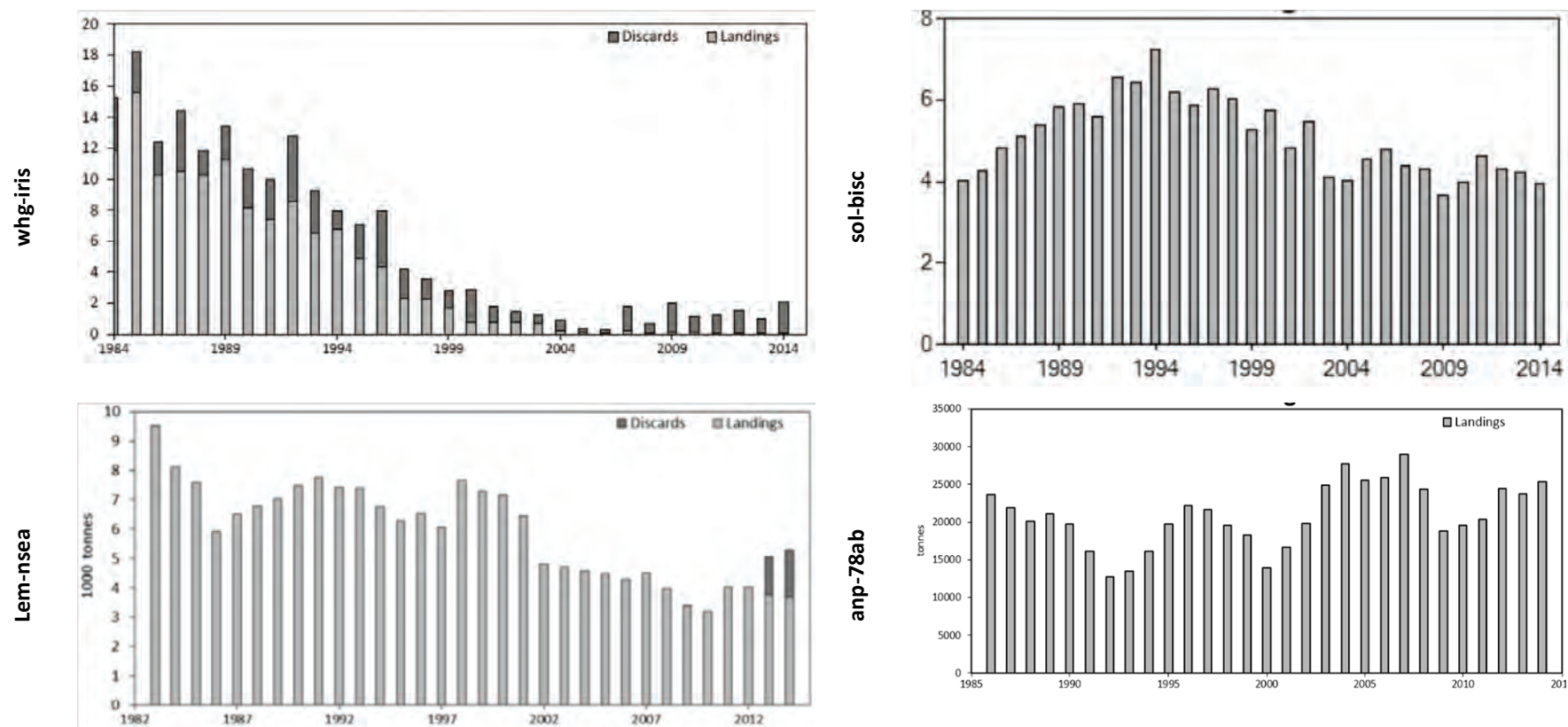


Figure 11.1 (continued). The evolution of fisheries' catches (landings and/or discards) over time and the inclusion of discards in stock assessments (ICES, 2015a). Fish stocks are described in detail in chapter 2: whg-iris: Whiting in the Irish Sea, lem-nsea: lemon sole in the North Sea, sol-bisc: sole in the Bay of Biscay, anp-78ab: anglerfish in the Celtic Sea and Bay of Biscay.

11.2 Details on discard data

The discard data from the first five databases were collected within the DCF framework, and follow the specifications introduced by this framework (EC, 2008a; 2008b; 2009b). Sampling should take place for instance in at least two fishing trips per quarter and metier, although data from smaller sample sizes may also have been used in practice (Vandemaele S., pers. comm.). Details on the number of realised trips, the fishing operations sampled, the number of species recorded and the parameters estimated by species can be found in the detailed reports linked to each database.

Data from the first data source, the STECF database, are described in detail in STECF (2013b, 2014a). This chapter focuses on a selection of the 2012 data which are summarised by management areas and/or special management regimes in Table 11.1 .

Table 11.1 The overlap between STECF management areas and ICES fishing areas (Figure 1.3).

STECF management area	ICES fishing area
Bal	Baltic Sea (Area 22 to 24; 25 to 28; 29 to 32)
3b2	Subarea IV (North Sea)
3a	Division IIIaS (Kattegat)
3b1	Division IIIaN (Skagerrak)
3b3	Division VIId (Eastern English Channel)
3c	Division VIIa (Irish Sea)
3d	Division VIa
IIb	Divisions VIIIc and IXa
IIc	Division VIle (Western English Channel)
WW	Western waters
BOB	Division VIIIab (Bay of Biscay)
Cel1	Divisions VIIbcefghjk
Cel2	Divisions VIIfg
FDFBAL	Fully Documented Fishery Baltic
FDFIIA	Fully Documented Fishery Cod Recovery Zone
FDFIIC	Fully Documented Fishery Western Channel

The information of the second data source, the ICES-database, was used to determine the advice to the EC on the Total Allowable Catches (TAC) (ICES, 2015a), which implies that discard data are provided by stock rather than by ICES Division.

The discards from the STECF- and ICES-database are not directly comparable due to areal differences, and should also be avoided as they result from different data calls and thus aggregations. The ICES data calls and the catch data used to provide stock assessments are uploaded through InterCatch (www.ices.dk) and detailed in the reports of the ICES Working Groups, i.e. area misreportings, uploaded data, data raising procedures and allocation to unsampled strata (ICES, 2015b, 2015c, 2015d).

Stocks were selected for evaluation in this study if

- A formal stock assessment was conducted to provide catch advice (for setting TACs).
- The species were important in Belgian fisheries, based on the landed biomass.

A first selection is based on the species and areas which are formally assessed by ICES and for which a TAC is set for Belgium. The species for which Belgium received a TAC in 2015¹¹ are: anglerfish (*Lophius sp.*), cod (*Gadus morhua*), Atlantic herring (*Clupea harengus*, hereafter herring), Atlantic horse mackerel (*Trachurus trachurus*, hereafter horse mackerel), Atlantic mackerel (*Scomber scombrus*, hereafter mackerel), brill (*Scophthalmus rhombus*), common dab (*Limanda limanda*), common sole (*Solea solea*, hereafter sole), European hake (*Merluccius merluccius*, hereafter called hake), plaice (*Pleuronectes platessa*), European sprat (*Sprattus sprattus*, hereafter sprat), haddock (*Melanogrammus aeglefinus*), lemon sole (*Microstomus kitt*), ling (*Molva molva*), megrim (*Lepidorhombus whiffiagonis*), Norway lobster (*Nephrops norvegicus*), pollack (*Pollachius pollachius*), redfish (*Sebastes sp.*), saithe (*Pollachius virens*), skates and rays (Rajidae), turbot (*Scophthalmus maximus*), whiting (*Merlangius merlangus*).

The TACs of sole and plaice are increased by exchanging TACs with other member state, accounting for a reduction in TAC of about 9000 tonnes herring, >1500 and 700 tonnes of sprat and mackerel, and about 650 tonnes of anglerfish in 2014. These TACs are reflected by the landed biomasses of Belgian fisheries, except for brown shrimp (*Crangon crangon*, hereafter called shrimp). The species composition of the landings is significantly different across regions (Figure 11.2; Figure 11.3), but regional differences were not addressed here. TACs are not set for shrimp fisheries. The most important species by landed biomass are in decreasing order: plaice (8449 tonnes), sole (3471 tonnes), molluscs and cephalopods (1915 tonnes), crustaceans including shrimp (1655 tonnes), cod (1263 tonnes), lemon sole (1137 tonnes), Rajidae (1041 tonnes), etc. Most of the landings result from the flatfish-directed beam trawl fishery, which is the main fishing metier in Belgium, besides the shrimp-directed beam trawlers.

¹¹ As fixed by Council Regulations (EU) No 1221/2014 of 10 November 2014, No 1367/2014 of 15 December 2014, No 2015/104 of 19 January 2015, and No 2015/106 of 19 January 2015.

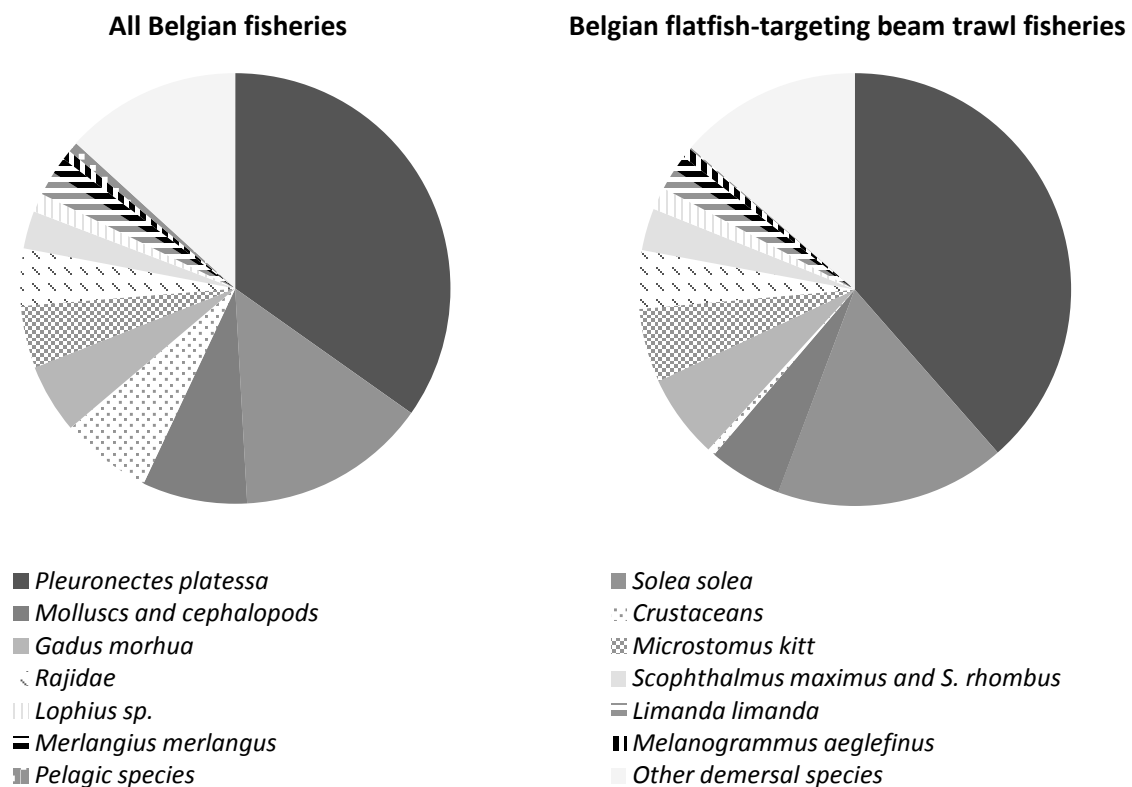


Figure 11.2. Species composition of the biomass landed by Belgian fisheries (left) and Belgian beam trawl fisheries targeting flatfish (right) in 2014: 24274 and 18761 tonnes respectively. Belgian fisheries are dominated by flatfish-targeting beam trawlers, although the species composition is also strongly influenced by shrimp beam trawlers, landing >1000 tonnes of shrimp in 2014.

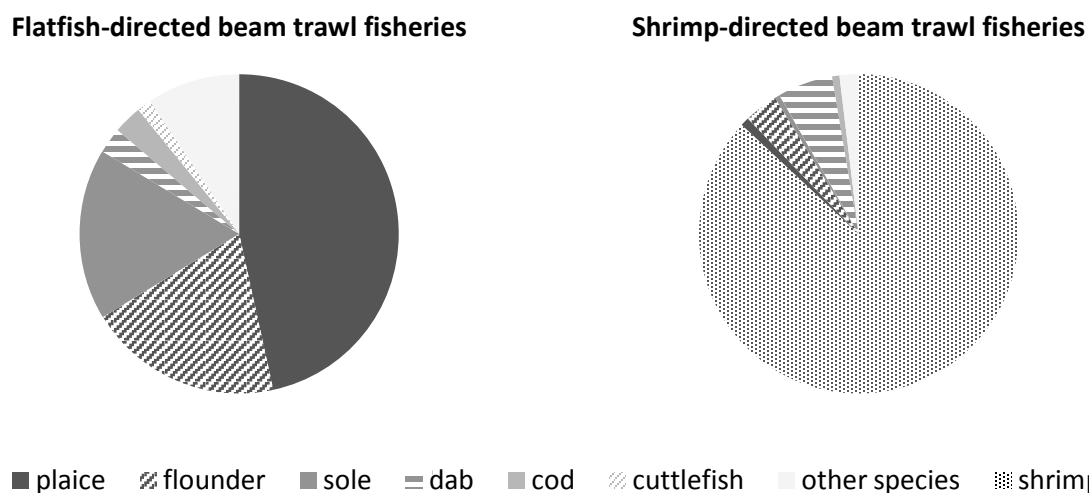


Figure 11.3 Species composition of the mean annual biomass landed extracted from the BPNS by the Belgian flatfish-directed beam trawl fishery (left) and the shrimp-directed fishery (right) (2012-2014): about 1400 and 980 tonnes respectively. Shrimp beam trawl fisheries contribute substantially to the landings of Belgian beam trawlers in the BPNS.

The stocks which relate to those species stem from different ICES fishing areas. Stocks that were formally assessed by ICES using the conventional ICES-methodologies (ICES, 2015a) are tabulated in Table 11.2. These stocks were the basis for the analyses in this study. Note that two species were excluded from the analysis despite their contribution to the landings (Figure 11.2):

- Brown shrimp (*Crangon crangon*) fisheries were not included as TACs are not set and a yearly, formal assessment is not conducted with the conventional ICES framework.
- Skates and rays were also excluded from this analysis despite their importance in the landings, but skate and ray species are not assessed in a similar way as other stocks. Assessments of elasmobranch species stocks suffer a lack of reliable data, which is partially due to species (mis)identification. Catch advice is provided biennially on the overall exploitation of the skate and ray assemblage and not on particular species. The advice is based on the approach for data-limited stocks (ICES, 2012a). Discarding is known to take place in fisheries catching elasmobranch assemblages (typically mixed fisheries such as beam trawling), but its inclusion in assessing the stock is clearly complex, given the specific approach that stock assessments of elasmobranchs require (ICES, 2014b).

In contrast, flounder (*Platichthys flesus*) was originally also excluded from the analysis, because it is not landed in high amounts by Belgian fishermen. However, as flounder constitutes 20% of the mean yearly landings in the Belgian Part of the North Sea (BPNS) in 2012-2014, this species was retained for further analysis (Figure 11.3).

The third, fourth and fifth data source in this chapter relate to national discard observer programmes of France, the Netherlands and Belgium respectively. The Ifremer discard observer programme covers the French fisheries and is described in detail in Cornou *et al.* (2013), Dubé *et al.* (2012), Fauconnet *et al.* (2011) and Fauconnet (2014). About 30 onboard observers covered 90% of the observations and conducted annually >2500 trips or >6000 days at sea. This sampling effort generally covered <2% of the trips and days at sea of the entire fleet, but occasionally is as high as about 20% of the fleet effort. The Dutch discard observer programme covers the Dutch beam trawler and seine fleet (Uhlmann *et al.*, 2013a). A total of 61 and 20 trips were sampled onboard large (> 221 kW) and small (\leq 221 kW) commercial beam trawlers respectively (80 - 99 mm codend), whereby two boxes of discards were sorted to species level in the laboratory for two hauls in each trip. Fish were treated separately from other 'non-fish' species. The sampling coverage was expressed in days at sea at accounted for 2 % of the large beam trawler effort and 2.7 % for eurocutters.

The ILVO discard observer programme covered the Belgian beam trawlers fleet and accounted for >40 trips and <5 observers in recent years. Sampling covers ~1% of all fishing effort (Vanellander *et al.*, 2015; Vandemaele *et al.*, in prep.).

Table 11.2 Abbreviations of important stocks (species and fishing area) for Belgian fisheries with an indication of the total biomass that is annually extracted by European fisheries (mean of 2012-2014).

Abbreviations of ICES stocks	Species	ICES Subareas and/or Divisions	Yearly fisheries catches (1000 tonnes)
ple-7h-k	Plaice	VIIh-k	0.2
anb/anp-78ab	Anglerfish	VII/VIIIab	45.8
sol-bisc*	Sole	VIIIab	4.3
sol-celt	Sole	VIIIfg	1.1
tur-nsea	Turbot	IV	2.9
bll-nsea	Brill	IV, IIIa, VIId	2.1
sol-iris	Sole	VIIa	0.2
sol-nsea	Sole	IV	14.7
sol-eche	Sole	VIIId	4.3
lem-nsea	Lemon sole	IV, IIIa, VIId	3.7
ple-nsea	Plaice	IV, IIIa	83.0
fle-nsea	Flounder	IV, IIIa	2.0
ple-eche	Plaice	VIIId	3.5
ple-celt	Plaice	VIIIfg	0.4
ple-iris	Plaice	VIIa	0.4
dab-nsea	Dab	IV, IIIa	6.1
had-346a	Haddock	IV, IIIa, VIa	40.3
cod-7e-k	Cod	VIIe-k	6.0
whg-7e-k	Whiting	VIIe-k	11.7
cod-347d	Cod	IV, IIIa, VIId	32.8
had-7b-k	Haddock	VIIb-k	13.9
whg-47d	Whiting	IV, VIId	18.4
cod-iris	Cod	VIIa	0.2
whg-iris	Whiting	VIIa	0.1

The sixth data source does not fall under the DCF. Catch data were not collected with the objective of sampling discards, but were collected to evaluate the numerical differences in catches between a conventional beam trawl and a beam trawl modified to reduce the discards.

The focus was primarily onto commercial species, as the gear modifications were designed to retain the landings of commercial species if fishermen are to use them. Species were arbitrarily categorized as commercial, based on Figure 11.2. This categorization is a qualitative evaluation of the commercial importance of species and reflects largely the division between commercial and non-commercial species, but is nevertheless artificial. Herring, for instance, was treated as a non-commercial species in the beam trawl trials.

Two commercial fishing trips were conducted by a flatfish-directed beam trawler (>221 kW) in the Celtic Sea (ICES Division VIIIf, one trip, 29 sampled hauls) and the English Channel (ICES Division VIIde, one trip, 22 sampled hauls). Trips were conducted in the first quarter of 2008 to test a square mesh codend and/or benthos release panel (Vandendriessche *et al.*, 2008). The discards of the commercial species were sorted from the non-commercial species, weighted and enumerated. The discards of non-commercial species were subsampled at least one 12L or in 40L buckets. The total catch was weighted as well as the subsample, which was used to raise the number of species to haul level. Data were collected by two observers.

Three fishing trips with a small beam trawler (≤ 221 kW) were conducted in the southern North Sea in the second quarter of 2007-2008 to test a T90 codend and/or benthos release panel (Depestele *et al.*, 2008b; 2008c). These data were complemented with catch data from a commercial 4m beam trawl, deployed from the RV 'Belgica' between the years 2004 - 2011. The beam trawl used onboard the RV 'Belgica' was rigged with a chain mat in contrast to the one from the commercial beam trawler, which used a limited number a tickler chains. The data of the RV 'Belgica' were combined with those from the commercial beam trawlers, because this increased the number of sampled hauls and the representativeness of the gear in the southern North Sea. Chain mats are commonly used by most commercial beam trawlers in the southern North Sea (Rijnsdorp *et al.*, 2008). A similar procedure was followed as with the commercial beam trawlers in ICES Division VIIde and VIIIf, although the data collected represent the catch rather than the discards, as most hauls were conducted by a RV rather than a commercial beam trawler. A total of 765 hauls were used to assess the catch composition in the western part of the southern North Sea (IVc West). The eastern part of the southern North Sea, comprising the BPNS, was also sampled by the RV 'Belgica' but limited to 7 hauls (Depestele *et al.*, 2008b, 2012; Van de Walle, 2012; www.mumm.ac.be). Data were collected by two observers onboard commercial fishing vessels or by a team of fishery scientists onboard the RV 'Belgica'.

Additionally, the non-commercial catch of twelve hauls conducted onboard the RV 'Belgica' were analysed in fractions of 12L in order to assess the accuracy of subsampling the non-commercial catches. Locations and gear specifications are described in detail in Van de Walle (2012).

11.3 Details on the recommendations to improve discard partitioning

11.3.1 Variability and uncertainties in seabird attraction

The high variation in the number of ship followers associated with fishing vessels cannot be easily linked to either natural occurrence of prey (Bertrand *et al.*, 2014; Certain *et al.*, 2011) or discards (Chapter 8). Down-scaling the spatial scale to investigate the interactions between seabirds and fisheries at high resolution may increase our insights (Torres *et al.*, 2013a; Torres *et al.*, 2013b), although the exclusive use of fine-scaled data has also its limitations (e.g. limited coverage of the population) and therefore a combination of scales is recommended (Croxall *et al.*, 2013). Tools were recently developed for both the evaluation of fine-scale movements of seabirds and fishing activities.

Fisheries' distributions at high resolution

The Vessel Monitoring System (VMS) was introduced to control whether activities of fishing vessels were taking place in areas where fishing is allowed (European Commission (EC), 2002). VMS data, however, are also a valuable source of information for scientists, as they allow, to a certain extent, to detect fishing activities of individual commercial fishing vessels >15m (Bastardie *et al.*, 2010; Hintzen *et al.*, 2010; Lee *et al.*, 2010). VMS data are based on GPS positions of individual vessels, which are combined with speed and heading at a minimum of every two hours. The locations of fishing are based on the assumption of differences between speed during fishing and non-fishing activities.

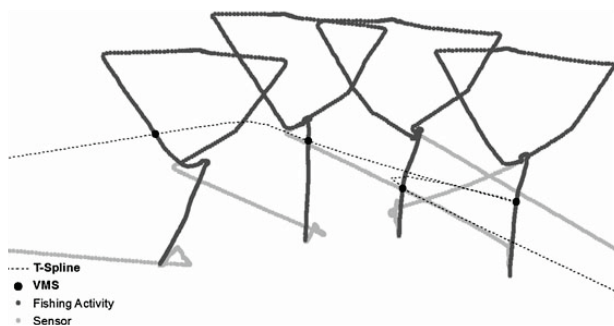


Figure 11.4 Fishing locations from VMS “pings” and 10 sec position records from a GPS linked to a Remote Electronic Monitoring system (dark grey for fishing, light grey for non-fishing) on a Scottish seine vessel. The dotted line gives a T-spline curve fitted through the VMS points, which would be the best estimate of the area impacted by the vessel in the absence of REM data (After Needle *et al.*, 2014).

The use of VMS data has three major disadvantages. First, only vessels >15m are recorded. Second, despite the generation of data at a frequency of 10 s, the VMS pings are only available every two hours to reduce the cost of satellite transmissions. The sparseness of the VMS pings implies that the accuracy of the trawl tracks is about 500m in comparison to GPS tracks from trawling vessels (Skaar *et al.*, 2011). Third, the detection of fishing activities is indirect, which are sufficient for mobile fisheries, but which are challenging for passive gears (Figure 11.4; Needle *et al.*, 2014).

Seabird activities at high resolution

Tracking studies have provided valuable information on the at-sea distribution of a small subset of scavenging seabirds. The major advantage of tracking studies is the capacity to provide information on an individual level (sex, age, body condition), as well as distribution information ranging from low to very high resolution over various temporal and spatial scales (Ponchon *et al.*, 2013). Different techniques are available (archival GPS loggers, remotely downloadable GPS loggers, etc.) that allow the detection of bird activities from GPS locations in three dimensions (geographical position as well as height), flight speed (Votier *et al.*, 2010), environmental characteristics such as temperature and air pressure (Shamoun-Baranes *et al.*, 2011), physiology such as heart rate (Bouten *et al.*, 2013) or from bird-borne cameras (Tremblay *et al.*, 2014; Votier *et al.*, 2013). The combination of these variables reveals at-sea behaviour of seabirds, including their foraging habits (potentially with for instance stable isotope analysis).

Association between fisheries and seabirds

A number of studies have combined the fine-scale movement patterns of individual seabirds with the VMS positions from fishing vessels (Bartumeus *et al.*, 2010). The number of seabirds which are equipped with tracking devices is generally low, but its potential to improve our insights the interactions is high. Lesser black-backed gulls, for instance, were associated with fishing vessels during the weekdays in the Wadden Sea, but occurred less frequent during the weekends (Tyson *et al.*, 2015). In another study, Tremblay *et al.* (2014) noted that seabirds locate their prey indirectly by detecting foraging conspecifics and other indicators of prey such as fishing vessels (a self-organizing mechanism, known as 'local enhancement'), while Bodey *et al.* (2014a) revealed that Gannet behaviour is influenced by fishing vessels up to distances of 11km. The limited attraction area of seabirds to a fishing vessel was also calculated in Chapter 8.

The large-scale distribution of fisheries discards and seabirds as presented in Chapter 8 was complemented with an index of seabird attraction. However, fishing vessels as well as seabirds are highly dynamic, and the large-scale overlap of these static layers may therefore show a biased view of the complex interactions between seabirds and fishing vessels. Interactions may be better understood at a high resolution, e.g. to understand which fishing métiers primarily link to seabird scavenging. The foraging behaviour of 10 Gannets was evaluated by GPS-tags and bird-borne cameras in the English Channel, Western approaches, Celtic and Irish Sea (Votier *et al.*, 2013). Beam and otter trawlers were responsible for >90% of the discards, and hence about 93% of encountered fishing vessels by the Gannets were trawlers. The distances covered in a foraging trip ranged between 20 and >100km with area-restricted search (ARS) behaviour in the vicinity of fishing vessels

(Figure 11.5). The VMS-data of Belgian beam trawl fisheries are available (Vanellander *et al.*, 2015), and Herring and Lesser black-backed gulls have been tagged with GPS-equipment, opening possibilities to investigate links between fishing metiers and gulls from Belgian colonies (Figure 11.6).

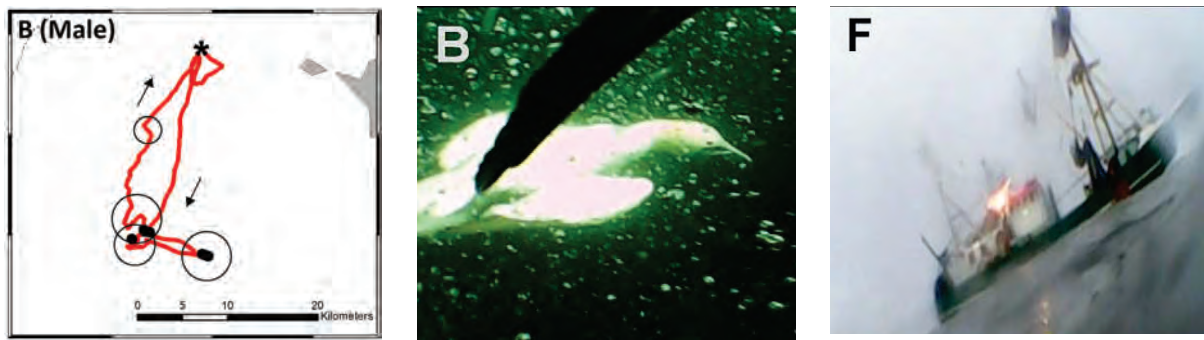


Figure 11.5 GPS-loggers (left panel) and bird-borne cameras (middle and right panel) indicate fishery-seabird interactions for a male Gannet. The foraging trip (red) was interrupted by ARS-behaviour (open circles) close to fishing vessels (black dots) or in an area without fishing vessels. Arrows show the direction of travel and an asterisk the location of the colony. After Votier *et al.* (2013).

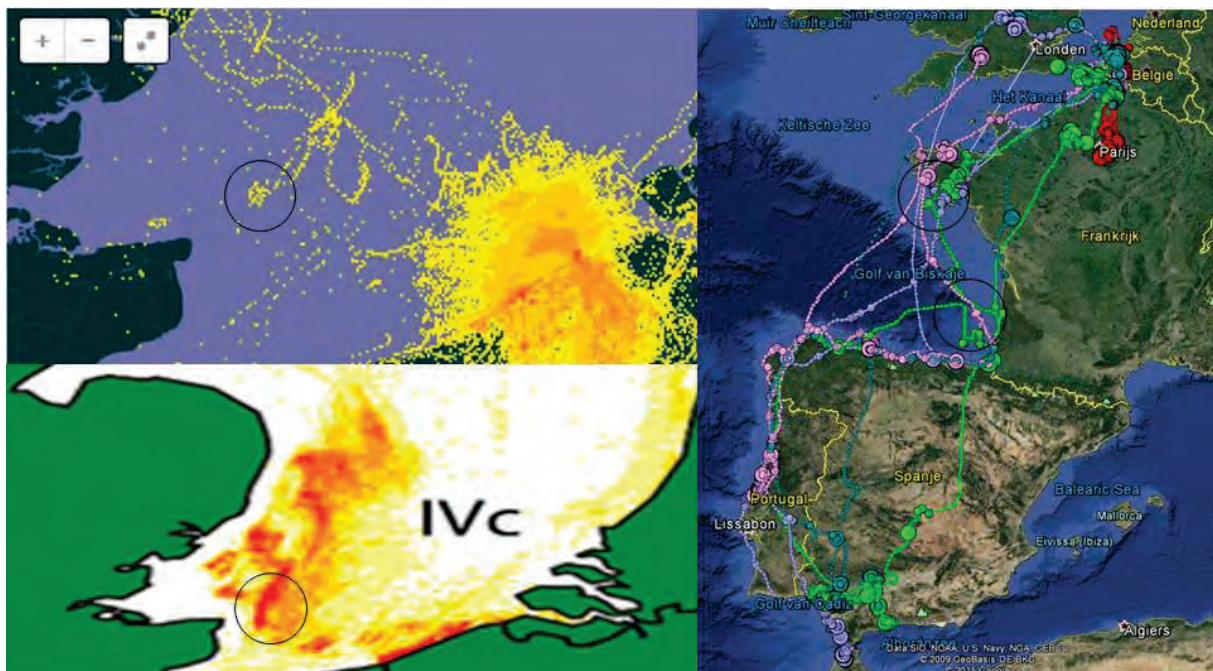


Figure 11.6 Low-quality screenshots of GPS-locations of Herring and Lesser black-backed gulls (upper left and right) indicate where gull species from Belgian colonies spent their time. The gulls mainly concentrate around Oostende and Zeebrugge, but offshore patterns are either remarkably concentrated or following straight line tracks. These patterns seem to occur where Belgian beam trawlers are operating (lower left panel). The gulls that cross the Bay of Biscay also seem to be located in the *Nephrops* fishing grounds or in the southern part of the Bay of Biscay, which coincides with locations where Gannets and Large gulls were feeding (Chapter 8). These un-analysed and anecdotic links indicate that there is scope for investigating the links between Belgian fishing vessels and gulls. After Vanellander *et al.* (2015) and www.lifewatch.be (8/5/2015).

11.3.2 *Variability and uncertainties in experimental discard consumption by seabirds*

Experimental discarding has been conducted on-board commercial fishing vessels (Hudson & Furness, 1989; Martínez-Abraín *et al.*, 2002) as well as research vessels (RV) (Camphuysen *et al.*, 1995; Garthe *et al.*, 1996). The main objective of the ship-based approach is to quantify the number of discarded items that are consumed by scavenging seabirds. While the discarding experiments on-board commercial vessels reflect commercial fishing practices, RV-based experiments have been significant in identifying the factors that influence discard consumption such as competition and kleptoparasitism (Chapter 8, Garthe & Hüppop, 1998b). The ship-based experiments have tended to focus on consumption rate of discards by seabirds, as well as examining patterns of variability due to the number of scavenging seabirds, composition of mix-species flocks (with relevance for intra- and inter-specific competition), the number and type of discarded items (e.g. fish with a rounded cross-body section are more attractive to seabirds than flatfish or invertebrates with protrusions), as well as size of discarded fish.

Variability in foraging success was extensively demonstrated in several, isolated studies as a dependent on a vast range of factors between and within species and individuals (e.g. Barrett *et al.*, 2002; Patrick & Weimerskirch, 2014; Votier *et al.*, 2013). Foraging success within species for instance relates to age and sex-specific differences (e.g. Lewis *et al.*, 2002; Navarro *et al.*, 2010; Stauss *et al.*, 2012; Weimerskirch *et al.*, 2009), while changing stages of food interest or feeding experience exemplify differences in foraging ecology within an individual over time (e.g. Garthe & Hüppop, 1998b; Greig *et al.*, 1983; Skórka & Wójcik, 2008). Additionally, but to our knowledge less frequently assessed, is the dependence of foraging success in relation to fisheries-specific characteristics. Flatfish-directed beam trawling for instance is conducted on a 24 h basis, while discarding of shrimp beam trawling or gill netting occurs mainly at night or dawn. The scavenging conditions vary greatly with day and night. Visibility for instance can reduce interspecific competition as some species, notably gannets and skuas, are assumed to be predominantly active during daylight hours (Garthe & Hüppop, 1993; Garthe *et al.*, 2012), while gulls exploit food equally well during day and night (Garthe & Hüppop, 1996). Similarities in day and night EDC of gulls are however contrasted by others which demonstrate that additional factors also play a role (e.g. Arcos & Oro, 2002b; Hill & Wassenberg, 2000; Sotillo, 2012). Another factor related to fishing métiers is the availability of discards to seabirds, which is determined by duration of the discarding process, catch composition and hence sinking rates (Hill & Wassenberg, 2000). Up till now, published experimental trials primarily focused on trawling. Studies have therefore assumed that feeding behaviour can be extrapolated across all fishing métiers (Camphuysen *et al.*, 1995: 92-94; Wagner & Boersma, 2012), although this may result in an over-estimation in some cases (Arcos & Oro, 2002b). While acknowledging these differences in

foraging success, data are lacking to account for all factors of variability when applying EDC to infer the number of consumed discards by seabirds on a population level (bird's view) or on a fleet level (fisheries' view). This clearly applies to published studies (Furness *et al.*, 2007; Garthe *et al.*, 1996; Kaiser & Hiddink, 2007) as well as Chapter 8.

As the driving factors proliferate spatially and temporally within and between populations, the deliberate choice of the spatial and/or temporal resolution and the number of replicates determine the extent of variability in the studied entities, even without explicitly identifying its causes. As such, differences in foraging success can be mediated by an overarching spatial and temporal approach and a sufficient number of replicates. The most extensive study of discard consumption for instance subdivided the North Sea in six regions (Garthe *et al.*, 1996). Variability in discard composition and logistic ease of sampling ('recognition of beam trawlers from other fishing vessels') were arbitrarily identified as drivers for spatial delineation. This *a priori* chosen spatial resolution mediated the local differences as for instance found between populations of Skuas in the NW regions in the North Sea (Votier *et al.*, 2008). Similar variability in EDC was found between local areas in the western Mediterranean (Oro & Ruiz, 1997), but evaluated on a lower spatial resolution in Furness *et al.*, 2007. While uncertainty is prone to these low-resolution estimates, EDC estimates were nevertheless used to generalize the total consumption of discards by seabirds. The uncertainty in estimates of discard consumption by seabirds is hence conditional on the assumption that EDC can be extrapolated to that spatial and/or temporal resolution, but also on other deficiencies such as the comparability between bird feeding experiments and commercial conditions (Stratoudakis, 1999). Despite these shortcomings, estimates created a valuable, generic picture of the importance of discards for seabirds.

Two major drivers of the high variability in the EDC are (1) ship followers' flock diversity and (2) discard diversity, because these two factors essentially determine which discards are available and which ones are easy to swallow by seabirds (Furness *et al.*, 2007). The influence of both factors is summarized below.

1. *Factors related to ship followers' diversity*

Foraging success varies by (1) seabird species, (2) flock size, (3) the presence of "other" seabird species and (4) interactions between them (ship follower's diversity).

Species level is an evident organisational level to discriminate between foraging successes (e.g. Furness *et al.*, 1992) and has consequently been the level of detail of most experimental studies. Feeding niche segregation, however, has been demonstrated from the individual level up to foraging

guilds and may justify other categorizations (Bicknell *et al.*, 2013; Bodey *et al.*, 2014b; Karnovsky *et al.*, 2012). Taxonomic and subsequent morphological differences are a primary driver of differences in EDC, but interactions between conspecifics and heterospecifics may induce large differences in EDC as well (Hunt & Furness, 1996), ranging from a limited effect of interference competition to complete exclusion. Seabirds do not capture discards to the proportional abundance of species behind fishing vessels. Yellow-legged gulls and Audouin's gulls (*Larus audouinii*) for instance avoid direct competition as yellow-legged gulls exhibit an aggressive feeding pattern which induces Audouin's gulls to feed at night or to consume smaller discards (Arcos *et al.*, 2001). This changes the EDC of Audouin's gulls and makes them more efficient in feeding on small discards (Martínez-Abraín *et al.*, 2002). Northern gannets displace herring gulls at fishing vessels and thereby alter discard consumption (Furness *et al.*, 1992; Figure 11.7). Camphuysen & Garthe (1997) stressed the influence of competition by illustrating that the roundfish EDC of northern fulmars (*Fulmarus glacialis*) varies between virtually zero to nearly 75%, depending on the number of fulmars behind a trawl and the presence of 'other' scavenging seabirds.

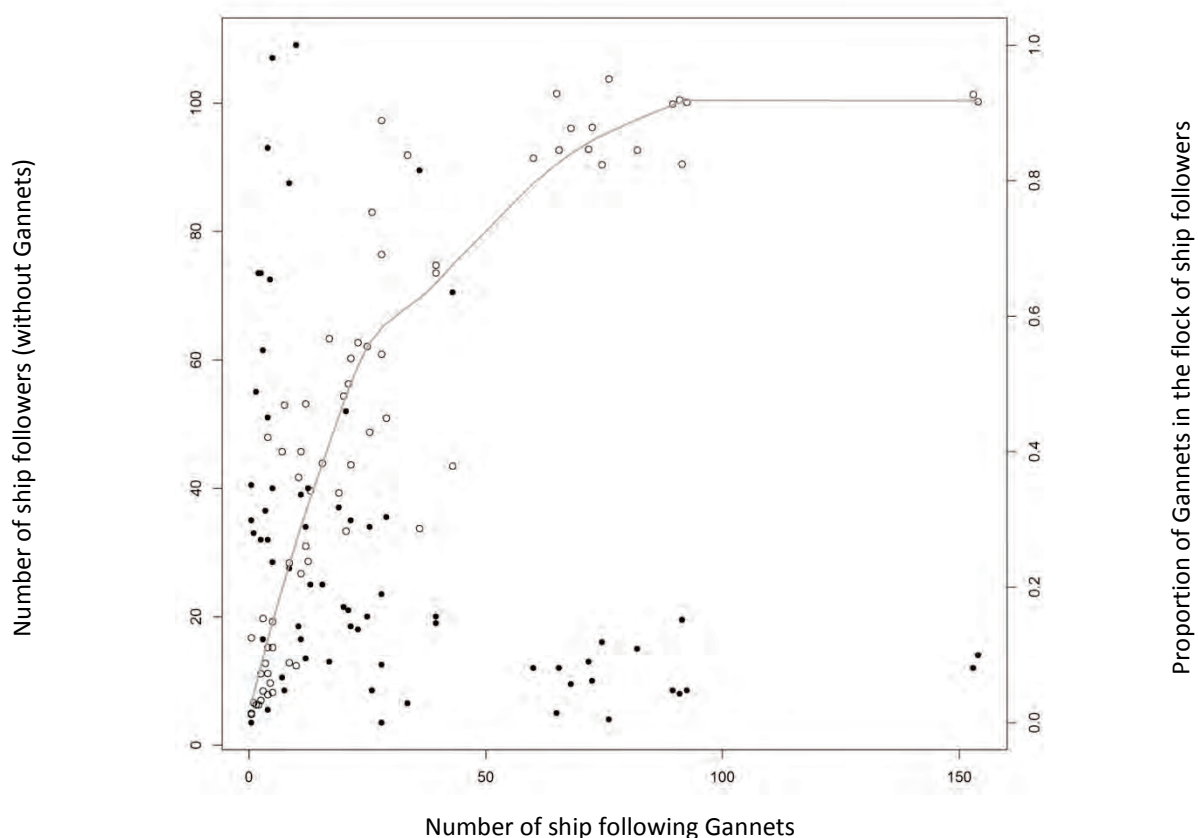


Figure 11.7 Competitive exclusion by ship following Gannets. The number of ship followers (black dots) decrease with increasing number of ship following Gannets. Conversely, the proportion of Gannets in the flock of ship followers increases (open grey circles). A smoother has been added to aid visualisation. Data are based on experimental trials described in Chapter 8.

Another example in the North Sea indicated that Herring gulls can be classified as offal specialist, while Lesser black-backed gulls are more successful in picking gurnards (Camphuysen, 1994; Camphuysen, 1995). The high competitive skills in Lesser black-backed gulls are likely due to manoeuvrability in mixed flocks, which is even more the case for kittiwakes and which affects the overall numbers of discards consumed (Schwemmer *et al.*, 2013). Competition between both gull species can also increase their consumption rates of flatfish discards (Camphuysen, 1994). The feeding success of sandwich terns in contrast decreased with an increasing number of competitors (Jodice *et al.*, 2011). Procellariidae use their diving capabilities when the number of ship followers is too high, and they scavenge only at the surface when the number of ship followers is low (Bugoni *et al.*, 2010). Other species increase their competitive abilities in larger flocks through kleptoparasitism (Gonzalez-Zevallos & Yorio, 2011; Votier *et al.*, 2008; Votier *et al.*, 2007) and alter EDC as time is invested in stealing prey items. This is how Greater black-backed gulls achieve the consumption of larger prey (Camphuysen, 1995; Camphuysen *et al.*, 1995). Less efficient scavenging behaviour occurs as well in seabirds which specialize in natural food items, but need to switch to discards when natural food is limiting (Louzao *et al.*, 2011; Schwemmer & Garthe, 2005). Dietary segregation mitigates potential competition between closely related organisms (Barger & Kitaysky, 2012; Thiebot *et al.*, 2011), and is complemented by spatial and temporal segregation among species with closely related feeding habits (Cama *et al.*, 2012; Cherel *et al.*, 2006; Jaeger *et al.*, 2010). Temporal segregation may also occur and is mostly related to breeding.

2. Factors related to discards' diversity

Discard consumption is evidently influenced by discard diversity as well: (1) discard type (Chapter 8), (2) discarded amounts, (3) discarded sizes, and (4) their interactions (discard composition or diversity). The sorting process on-board fishing vessels and other factors related to discarding practices (e.g. duration of discarding, location of discharging material in relation to the vessel, steaming speed of the vessel during discarding) affect the availability of discards as well, although only very few studies have focused on these matters (Furness *et al.*, 2007; Gilman *et al.*, 2013; Pierre *et al.*, 2012a; Pierre *et al.*, 2012b).

The number of discards that is offered at once has a substantial influence on EDC. A minimum availability triggers highly competitive feeding conditions at fishing vessels, including kleptoparasitic events (Garthe & Hüppop, 1998b; Votier *et al.*, 2004) and may lead to inflated EDC estimates. In contrast, a high number of discarded individuals (e.g. discard slippage in pelagic fisheries) at once may lead to decreased EDC. The variability of EDC in function of different quantities of discards offered are only limitedly investigated, but may be highly relevant when comparing EDC-estimates

between fishing metiers. The effect of the number of discards offered in relation to the numbers consumed has only been investigated for small numbers of discards, even though fisheries that are discharging large quantities at once are also significant sources of food for certain seabird species (e.g. purse seiners, Arcos & Oro, 2002b). The amount of discards offered was investigated to test the protocol of ship-based discard experiments (Garthe & Hüppop, 1998a). Garthe & Hüppop (1998a) quantified the difference between discarding individual organisms (single-item experiment) and multiple items at once or in a steady trickle over a short time span (multi-item experiment). The authors found clear differences in EDC for Kittiwakes and northern fulmars, but not for northern Gannets. The effect of the number of discards offered by unit of time was also tested for northern Gannets during the trials in the Bay of Biscay (unpublished data, Chapter 8). The consumption of discarded roundfish was lower when 50 fish were discarded at once in comparison to the consumption rate when 2.4 fishes were discarded per second (Figure 11.8). The differences were more pronounced for small fish (~10-20cm) than for larger fish (~20-30cm), indicating a preference of Gannets for larger fish (mostly horse mackerel). No differences between EDC boarfish were expected as EDC was low (<5%). The number of experiments was limited.

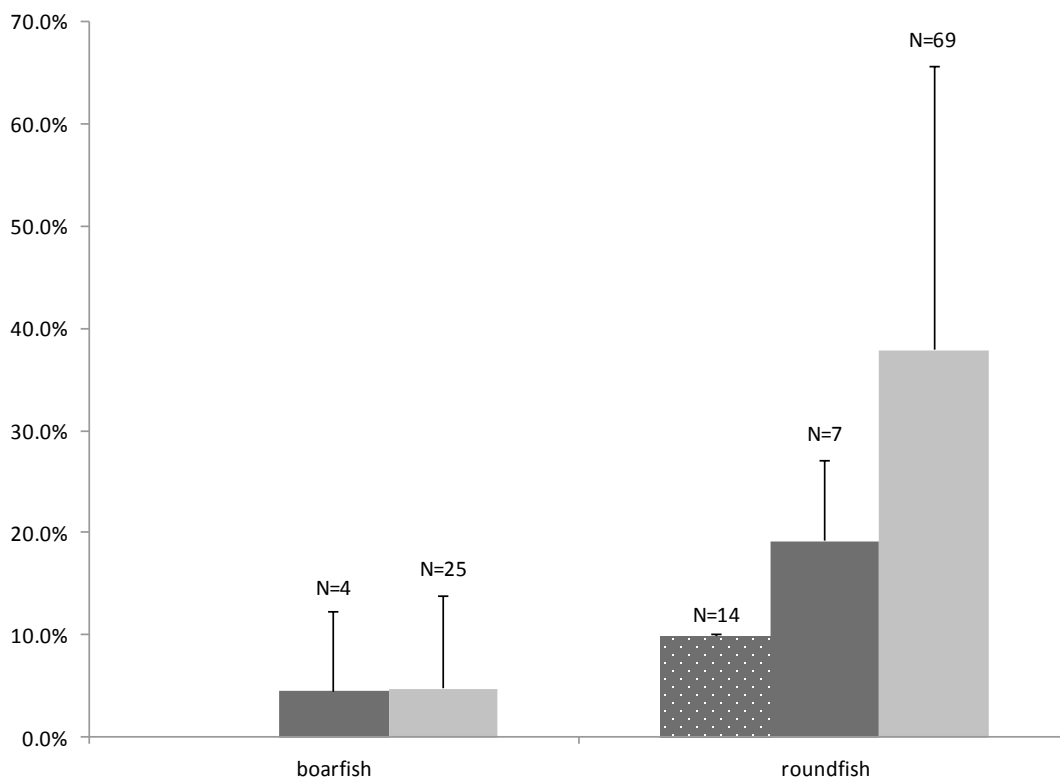


Figure 11.8 Mean Experimental Discard Consumption (+SD) by northern Gannets in the Bay of Biscay for discards discharges at once (dark grey) or in a steady trickle (light grey). Roundfish species included *Argentina spp*, *Melanogrammus aeglefinus*, *Merluccius merluccius*, *Micromesistius poutassou*, *Scomber scombrus*, *Trachurus trachurus*, *Trisopterus spp*, *Sebastes spp*. Roundfish discarded at once varied between small (dotted dark grey bar) and large (solid dark grey bar) individuals.

Garthe & Hüppop (1998a) proposed a correction factor to accommodate for the differences between experimental methodologies, but none of the peer-reviewed papers citing the authors (Web of Science, 6 February 2014) have followed this recommendation. Instead, studies discarded at least 30 items to account for possible biases or used a steady trickle of multiple discards to reflect commercial practices (Catchpole *et al.*, 2006; Jodice *et al.*, 2011, Chapter 8). While this methodology generally decreases the EDC in comparison to single-item experiments, it is currently unclear whether the provision of discards in these multi-item experiments reflects commercial fishing practices, e.g. maceration of discards into small chunks or providing a steady trickle of whole fish during a period of 20min for beam trawlers (Chapter 5, Furness *et al.*, 2007).

Differences between discarded species and sizes were also highlighted as an important factor determining EDC-estimates in western Mediterranean studies (Arcos & Oro, 2002a; Arcos *et al.*, 2001; Oro & Ruiz, 1997). The size distribution of discarded fish consumed peaks in the range between 10 and 20 cm for seabird communities in the Western Mediterranean sea, and differed by species. Audouin's gull selected significantly smaller discards compared with the larger yellow-legged gull (Arcos *et al.*, 2001). The feeding preferences of seabirds for instance were investigated in the North Sea (Camphuysen *et al.*, 1995; Garthe *et al.*, 1996), revealing that the number of flatfish consumed rapidly decrease within increasing sizes, while this is less so for roundfish (Figure 11.9). The number of flatfish consumed for instance can vary between 5% in whitefish trawlers of Shetland (Hudson & Furness, 1988) and 41% in shrimp beam trawling in the Wadden Sea (Walter & Becker, 1997). Mesh sizes of beam trawlers for shrimp (*Crangon crangon*) are smaller than for whitefish trawlers, and although many additional factors may have contributed to the difference EDC, discarded size is a likely important contributor.

Discards vary considerably in discarded amounts, types (or species) and sizes. The interactions between those factors may be an important driver of differences in EDC between fishing metiers. An equal amount of discards with a higher ratio of flatfish to roundfish for instance will induce a higher competition between bird species for roundfish. The discard composition of pelagic trawlers may, in this sense, be more suitable to scavenging seabirds than those of demersal trawlers (Figure 11.10). Also, the overlap in preference for medium-sized discards from Audouin's gulls and yellow-legged gulls was illustrated to induce alternative foraging techniques such as the selection of smaller prey items in Audouin's gulls (Arcos *et al.*, 2001). While the effect of each factor individually is well understood, this is less so for their interactions, complicating extrapolations of these findings across fishing metiers.

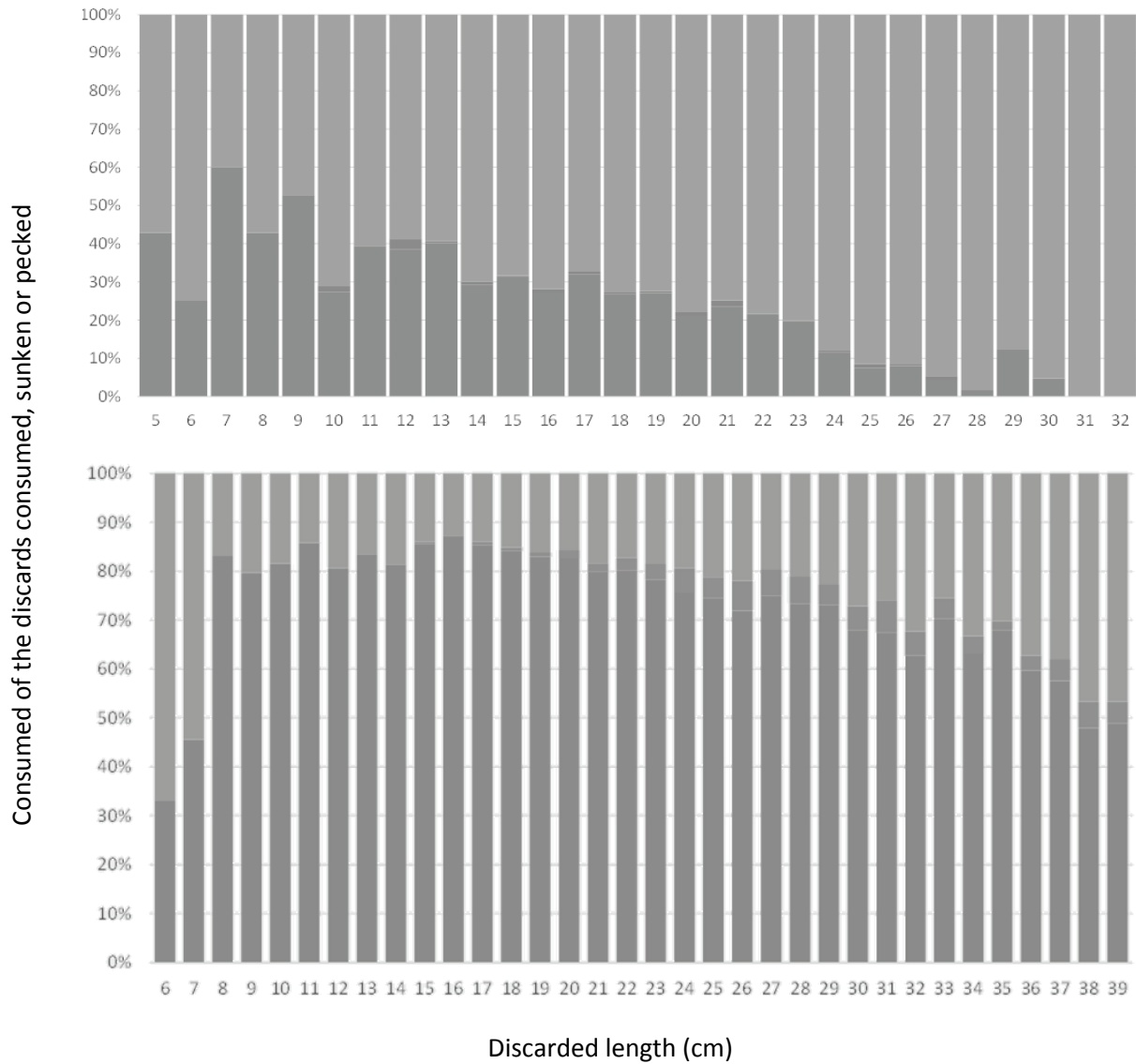


Figure 11.9 Percentage of flatfish (upper, N=6,422) and roundfish (lower panel, N=44,358) consumed by length class (cm) in the North Sea. The lower part of the bars indicate the consumed discards (dark grey), upper parts indicate sunken discards (light grey), while middle parts are discards that were pecked. The figure was kindly provided by Kees Camphuysen during the ICES Workshop on Methods for Estimating Discard Survival 2 (ICES, 2015b). Details can be found by species in Camphuysen *et al.* (1995).

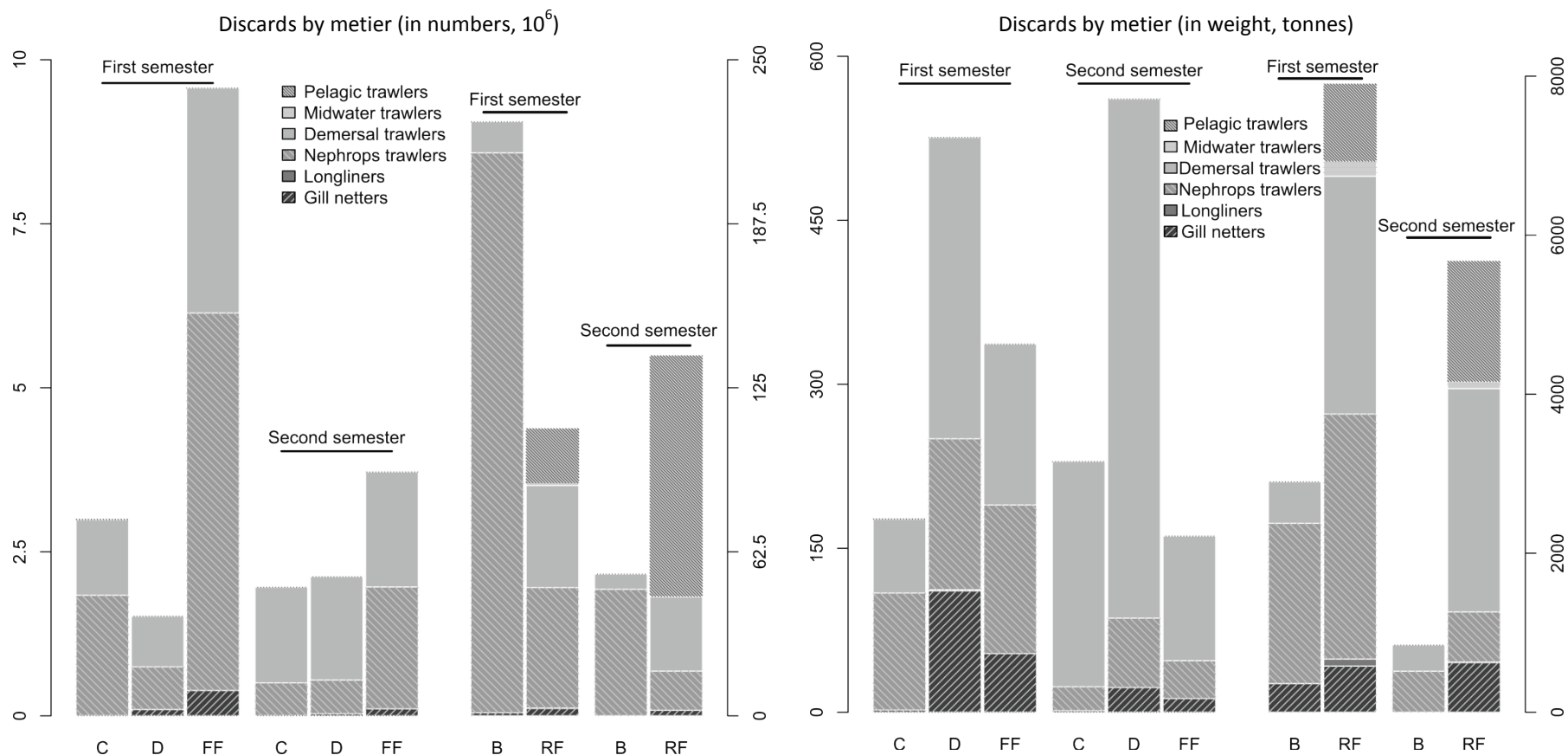


Figure 11.10 Discards by the six main fishing metiers in the Bay of Biscay. The total number of discards (left) is expressed in million number of discards (Y-axis). The total numbers of cephalopod (C), depressiform fishes (D) and flatfish (FF) discards respond to the primary axis; benthic invertebrates (B) and roundfish (RF) to the secondary axis. The total weight of discards (right) are expressed in tonnes. The primary axis relates to C, D and FF; while B and RF were reflected in the second axis.

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Epilogue

*Rather the flight of the bird passing and leaving no trace
Than creatures passing, leaving tracks on the ground.
The bird goes by and forgets, which is as it should be.
The creature, no longer there, and so, perfectly useless,
Shows it was there — also perfectly useless.*

*Remembering betrays Nature,
Because yesterday's Nature is not Nature,
What's past is nothing and remembering is not seeing.*

Fly, bird, fly away; teach me to disappear!

Fernando Pessoa

Acknowledgments

Nu het dan eindelijk mag, blijkt het moeilijker dan verwacht! Het is geen evidentie om elkeen zijn of haar poëtische plaatsje te mogen geven in de bedanking van deze thesis. Ik zou me bot kunnen vieren met een opsomming van alle personen die tot dit werk hebben bijgedragen, maar in plaats van alle namen over m'n tong te laten rollen, probeer ik mij te beperken tot een aantal mensen die net dat tikkeltje extra gaven waar ik nood aan had en die mij, nu, bij het schrijven te binnen schieten. Wie zich aangesproken voelt, maar van een kale zoektocht terugkeert, nodig ik uit om mij aan te spreken. Ik ben er zeker van dat deze thesis, laat staan de bedanking, niet het einde hoeft te vormen van onze enthousiasmerende samenwerkingen, en dat ik je hiervoor gepast te woord kan staan!

De bereidwilligheid van Hans (Polet) om mij te ondersteunen in het doctoreren verdient een ereplaats. Zonder zijn draagvlak zou ik in mijn hoedanigheid van onderzoeker aan het ILVO nooit de kans hebben gekregen om nog maar te kunnen denken aan doctoreren. Uiteraard ben ik hem vooral dankbaar voor het delen van de vele inzichten in de problematiek van de zeevisserij, als voor het open staan van zijn platgelopen deur. Hans, laten we erover waken dat pittig inhoudelijke discussies zich verder blijven opstapelen.

Het vertrouwen van Magda (Vincx) was insgelijks cruciaal. Subtiel, maar kordaat werd ik van een vrijgevochten vakgroep-bezoeker tot een doctorandus met academische realiteit gebracht. Voetjes op de grond, data in de hand en neus in de juiste richting. Meermaals kreeg ik op het juiste moment de juiste insteek om terug te keren naar de essentie, het hoofd koel te houden en met geloof in gedegen finaliteit af te ronden waar het in mijn gedachten nog maar pas begon. Het krijgen van een gerust vertrouwen in de toekomst deed me deugd. Magda, het zal een oprecht gemis vormen in het vervolg van mijn wetenschappelijke activiteitenpalet, maar de ervaring draag ik waardig mee.

Dat wetenschappelijk activiteitenpalet werd eveneens omvangrijk bijgekleurd door 'Techniek' met haar wisselende bezetting en haar vaste rotsen in de branding. Deze groep vormde de bakermat van waaruit ik mocht exploreren en experimenteren. Waar het in die pre-doctorale jaren met voorzichtige stapjes en een enge focus begon, kon de verruiming zich na enkele jaren met rasse schreden laten voelen. Het is een dankbaar gegeven te mogen ageren in de kritische massa van de groep (Bart, Hendrik, Els, etc.), de degelijk geaarde praktische ondersteuning (Fernand, Eddy, Christian, Kevin, etc.), haar zin voor visserij-realiteit en haar open blik naar andere

onderzoeksdomeinen. Laat die stevige verankering een springplank zijn naar een boeiend vervolg dat ik met nog meer plezier en nog meer openheid verder zet...

En dat dat inzetten op verder gelegen thema's en openheid belangrijk mag blijken, toonden reeds de vele samenwerkingen met andere onderzoeksgroepen. In het WAKO-project bijvoorbeeld mocht ik proeven van de inzichten van onder meer benthologische en ornithologische meerwaarde-zoekers en mocht ik ervaren hoe wij samen onze krachten trachten te bundelen om vanuit onze verschillende achtergronden de complexe visserij-thematiek door nieuwe vaarwaters trachten te loodsen. Het genoeg was geheel aan mijn zijde, en ik hoop dat jullie, Eric (Stienen), Wouter (Courtens), Steven (Degraer), Kris (Hostens) *et al.*, er danig van genoten hebben dat wij op het gepaste moment hier nog een vervolg aan kunnen breien. Hetzelfde geldt voor andere samenwerkingsverbanden tussen mensen van onderzoeksgroepen binnen het ILVO en er buiten (Marbiol, OD Nature, VLIZ, etc.). Het zijn jammer genoeg te veel initiatieven om op te sommen.


De stimulerende enthousiasmering van een ietwat breder kader mocht ik ook internationaal voelen door losse contacten en hechtere relaties met buitenlandse lotgenoten. De ICES Werkgroep voor ecosysteemeffecten van visserij (WGECO) prikkelde me meermaals en bracht me bijvoorbeeld tot concrete samenwerkingsverbanden met Marie-Joëlle Rochet. *L'expérience à Nantes m'a beaucoup plu et m'a particulièrement aidé à avancer d'une manière plus concrète et certainement plus efficace (R). La porte est ouverte pour une nouvelle coopération plus avancée dans le future proche! Les fondations sont construites. The myriad of the collaborative work in Nantes and other international fora cannot be underestimated. Symposia, hands-on statistical help (Hugues Benoît), collaborative work settings (Dominic Rihan, Leonie Robinson, Barry O'Neill, etc.), stimulating and thought-provoking discussions (Dave Reid...), EU-projects (Adriaan Rijnsdorp, Lorna Teal...) and so on, they've all added up to a pleasant nursery ground for fisheries' science and I am very grateful to be a part of this small but excited community.*

Evenzeer ben ik vereerd dat de leden van de lees- en examencommissie mijn uitnodiging tot deelname aan de verdediging hebben aanvaard. Een boeiende en levendige evaluatie van deze thesis is mij dierbaar en mag de poorten openen naar nieuwe en verfrissende toekomstideeën.

Het wetenschappelijke luik verbergt echter een resem aan logistieke ondersteuning. Naast kleine bedankinkjes of een appreciërend schouderklopje wordt de praktische kant veelal snel vergeten, en daarom wil ik elkeen die wetenschappelijke experimenten ondersteunt nog eens extra op het hart drukken hoe hard wij jullie nodig hebben. Ik denk hierbij onder meer aan de bemanning van het onderzoekingsvaartuig 'de Simon Stevin', die meerdere malen het onderste uit de kan heeft gehaald om de 'nagal' flexibele planning van de Benthis-experimenten te helpen uitvoeren, de bemanning

van de Belgica die het plezier van schijtende meeuwen moest genieten en al die zeelieden die de fragiele benen van mij en andere wetenschappers staande hielden en houden (vissers van O89, N95, O369, O554, SCH18, TX43, TX29, RV Thalassa, Z85, O32, etc.). Het eigenlijk werk gebeurt op zee, en er bestaat geen onderzoek zonder zij die de machtige wateren trotseren en daarbij open staan voor de zotten ideeën van terrestrisch landrotten.

Wetenschap zou mij ook niets betekend hebben zonder de menselijke kant van het totaalplaatje. De onrechtstreekse steun van mijn ouders, familie, en vrienden is onschatbaar. Ik hoef het hier niet op papier te zetten hoezeer ik ervan geniet als iemand openstaat om de dagdagelijkse beslommeringen van wetenschappelijk werk te aanhoren of meegaat in mijn enthousiasme naar een zoektocht tot verduurzaming. Ik hoef jullie niet zwart op wit te benoemen. Jullie weten hoezeer ik jullie ondersteuning heb geapprecieerd en nog steeds waardeer. Merci!



8 juli 2015, Gent.

About the author

Research interests and expertise

I was trained as a bio-engineer in forest and land management at Ghent University, though my research interest rapidly diverged from forest management and river habitat modelling into the sustainable use of natural resources at sea.

Being employed at the Institute for Agricultural and Fisheries Research from 2004 onwards, my primary attention was directed to fishing gear technology research. How can the target species of the Belgian fisheries be sustainably harvested through the use of beam trawls? How can we modify the gear to reduce discards and benthic impacts? The possibility to replace beam trawlers by gill or trammel netters was another important research focus, primarily due to the rising fuel prices in the 2008 period. While fellow scientists investigated the changes in profitability from gear conversion, my role was to evaluate the direct ecological effects of the gear shift.

The intensity of my studies on the direct ecological effects of beam trawling and/or a shift to trammel netting gradually increased and became my main field of research. Direct interactions were related to the catch including discards or physical contact with the seabed and its inhabitants. Several research questions have popped up ever since. The move towards the ecosystem approach of fisheries management likely requires aspirations on the secondary effects of the direct, ecological impacts of the gear. What is the fate of the discards? Trawling the seabed induces mortality in the tow path and alterations to the geophysico-chemical interface. Who profits from the discarded and dead or damaged organisms in the trawl path? Are the habitats of these animals impaired or in contrast, may they proliferate? These secondary effects (following the direct impacts) are part of my current research focus.

While the nearby future is expected to provide better insight into these secondary effects, it is one of my aspirations to link the range of direct and secondary effects to each other and evaluate how these interactions may alter the ecosystem in a one-event, stationary way or in a dynamic sequence of fishing events. Being at the forefront of gear innovations implies that changes in direct effects from changes in fishing practices may rapidly be detected. By keeping an eye on the provisioned impacts and propagations throughout the system, we may be able to provide well-founded advice that may help us to choose the directionality towards a sustainable use of marine resources. It is my ambition to further contribute to these research topics in one way or another.

Research projects

- Benthic Ecosystem Fisheries Impact Studies (Benthis, EU-FP7 project). *Aim:* Benthis studies the impacts of fishing on benthic ecosystems and will provide the science base to assess the impact of current fishing practices (2012-2017, contribution: Quantifying food subsidies to the benthos due to discards and consequences for ecosystems, assess trawling impacts of current and alternative gears in the North Sea case study).
- An integrated impact assessment of trammel net and beam trawl fisheries (WAKO-II, Belgian Federal Science Policy) *Aim:* (1) Fill out knowledge gaps in our understanding of trammel net and beam effects in the Belgian Part of the North Sea (identified in WAKO) and (2) Set up an integrative approach to assess ecosystem effects of fishing effects through sensitivity assessment. (2008-2012, project coordinator).
- Impact assessment of the effects of a selected range of fishing gears in the North Sea. Financed by Stichting de Noordzee –WWF, the Netherlands. 122 pp. (2009-2010, desk study).
- Studies and pilot projects for carrying out the common fisheries policy. LOT 3, scientific advice concerning the impact of the gears used to catch plaice and sole. (EU-Open call for tenders FISH/2007/7). *Aim:* Evaluate the impact of fishing gears currently used to catch plaice and sole in the North Sea and investigate (and if appropriate, recommend) the use of alternative fishing gears for the fisheries concerned. (2007-2010, contribution: review of environmental impacts, assistance in project coordination).
- Evaluation of environmental impact of beam trawls and trammel nets in the Belgian Part of the North Sea (WAKO, EU-FIFG, Flemish Government)(2006-2008, project coordinator).
- ‘Development of fishing gears with reduced effects on the environment’ (DEGREE, EU-FP6 project). *Aim:* Development of new fishing methods with a reduced impact on benthic habitats, quantifying the potential reduction of physical impact and the adverse effects on benthic communities, socio-economic evaluation of proposed gear modifications(2006-2009, contribution: experimental sea trials: beam trawl modifications).
- ‘Alternative Fisheries’ (PAV, EU-FIFG/Flemish Government). *Aim:* Development and pilot trials of alternative (passive) fishing methods (2005-2006, contribution: desk study and experimental sea trials: hand lining, jigging and long lining).
- ‘Innovation centre Sustainable and Ecological Fisheries’ (IDEV, EU-FIFG/Flemish Government). *Aim:* inventory and selection of sustainable and ecological fishing methods and development of gear modifications to reduce benthic impacts of beam trawls (2004-2007, contribution: desk study on fishing gears, experimental sea trials: beam trawl modifications).

Research affiliations

- ICES Working Group on Ecosystem effects of fishing (WGECO) (2007-2015)
- ICES Workshop on Methods for Estimating Discard Survival (WKMEDS) (2014-2015)
- ICES Advice Drafting Group on Vulnerable marine Ecosystems (ADGVME) (2012)
- ICES Workshops of 'Fisheries Measures in Protected Areas' project (FIMPAS) (2010-2011)
- ICES Workshop on Discard Raising Procedures (WKDRP) (2007)
- ICES Working Group on Fishing Technology and Fish Behaviour (WGFTFB) (2005-2008)
- ICES Study Group on the Development of Fish Pots for Commercial Fisheries and Survey Purposes (SGPOT) (2005-2007)
- Taskforce for LCIA programme LCIA-2 Natural resources and land use, UNEP/SETAC Life Cycle Initiative (2005-2006)

Research campaigns

~30 research cruises were conducted on-board national and international Research Vessels (RV 'Belgica', RV 'Clupea', RV 'Simon Stevin', RV 'Thalassa', RV 'Zeeleeuw'), and ~10 cruises on-board commercial fishing vessels (beam trawl, pulse trawl, otter trawl and trammel nets). Most cruises were conducted in the southern North Sea, besides the Bay of Biscay, the Irish and Celtic Sea and the Moray of Firth).

Research fellowships

- Fellowship 'instream habitat program' Cornell University, USA (MSc thesis) (1 month, 2003, Supervisor: Prof. dr. Piotr Parasiewicz)
- Fellowship 'Biological Traits Analysis' Liverpool University, UK (1 week, 2008, Supervisor: Dr. Leonie Robinson)
- Fellowship 'Euromarine mobility program' Ifremer EMH Nantes, France (3 months, 2012, Supervisor: Dr. Marie-Joëlle Rochet)

Supervision of BSc and MSc students

- Desmet, C. 2015. Which benthivorous fishes feed greedily after bottom trawling? Master of Science in Marine Biodiversity and Conservation Thesis. Ghent University; Faculty of Science: Ghent, Belgium. 25pp. (promotor: Prof. Dr. M. Vincx, supervisors: Jochen Depestele, Jozefien Derweduwen).
- Goossens, J. 2014. De soortselectiviteit van boomkorren vergroten: hoe vangen we de doelsoort, tong (*Solea solea*), met zo weinig mogelijk teruggooi? Bachelor in Biology. Ghent University; Faculty of Science: Ghent, Belgium. 25pp. (promotor: Prof. Dr. M. Vincx, supervisor: Jochen Depestele, Heleen Lenoir). 22pp.
- Blanc, N. 2013. Discards as food for scavenging seabirds in the southern North Sea. Master of Science in Marine Biodiversity and Conservation Thesis. Ghent University; Faculty of Science: Ghent, Belgium. 25pp. (promotor: Prof. Dr. M. Vincx, supervisor: Jochen Depestele) 30pp.
- Gorim, D.A. 2012. The potentials of gear modifications to reduce discards of beam trawl in the southern North Sea. Master of Science in Marine and Lacustrine Sciences and Management (Oceans and Lakes): Ghent, Belgium. 60pp. (promotor: Prof. Dr. M. Vincx, supervisor: Jochen Depestele).
- Maleš, J. 2012. The potential of gear modifications to improve species selectivity of beam trawls. Master of Science in Marine Biodiversity and Conservation Thesis. Ghent University; Faculty of Science: Ghent, Belgium. 25pp. (promotor: Prof. Dr. M. Vincx, supervisor: Jochen Depestele).
- Sotillo, A. 2012. Consumption of fishery discards by seabirds in the southern North Sea. Master of Science in Marine Biodiversity and Conservation Thesis. Ghent University; Faculty of Science: Ghent, Belgium. 51pp. (promotor: Prof. Dr. M. Vincx, supervisors: Wouter Courtens, Jochen Depestele).
- Van de Walle, S. 2012. Subsampling accuracy in commercial beam trawl catches. Master of Science in Marine Biodiversity and Conservation Thesis. Ghent University; Faculty of Science: Ghent, Belgium. 25pp. (promotor: Prof. Dr. M. Vincx, supervisor: Jochen Depestele).
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Popular publications

Research results are regularly distributed by popular magazines of the fishing industry (primarily ‘de Rederscentrale’ and ‘Visserijnieuws’) and occasionally by the regular and social media.

The return of unwanted catches to the sea is known as discarding. The composition and amounts of fisheries' discards are difficult to quantify, and our understanding of the fate of the returned catches is fragmented. The ecological role of discarded organisms in the marine ecosystem is by consequence also poorly understood.

This thesis aimed at resolving this gap by investigating the fate of discards from several perspectives: the human (stock) as well as the ecosystem perspective. Why do men discard part of the catch? What happens to the discarded organisms? Are they able to survive and overcome of the stressors from the catching process? Do they serve as food for scavengers? This thesis is a step towards an increased understanding of the role of discards in the marine ecosystem and the sustainable exploitation of resources by marine fisheries.

The thesis has been submitted in partial fulfilment of the requirements for the degree of Doctor of Science: Biology

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ISBN: 978-90-5989-816-5



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