# Seafood-waste disposal at sea – a scientific review.

Report to The Sea Fish Industry Authority

Institute of Estuarine and Coastal Studies
University of Hull

6 May, 2005

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# Sea Fish Industry Authority

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| For and on behalf of the Institute of Estuarine and Coastal Studies | |
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| Approved by: | |
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## EXECUTIVE SUMMARY

Due to the recent tightening of regulations surrounding the disposal of seafood processing waste to landfill, fish and shellfish processors are now facing a rise in the cost and difficulty of waste disposal. This is of particular concern in remote areas where alternative uses (e.g. fishmeal) are neither accessible nor economically viable and therefore, cost effective and environmentally-sound solutions to the disposal of this waste need to be found. The present report examines the potential for disposal at sea, together with the likely impacts and advice on the selection of appropriate sites for disposal. A number of case studies where the disposal of seafood processing waste at sea occurs are also reviewed in order to demonstrate the effects of this type of disposal. There is a general lack of information regarding the dumping of seafood waste at sea although, given the organic nature of the waste, the effects are expected to be similar to those associated with the disposal of sewage sludge, organically-enriched dredged material and fish farm waste.

There have been cases in the past where adverse impacts on the marine environment have been attributed to waste disposal at sea, especially in relation to the disposal of sewage sludge and fish farm waste. However, where disposal is carried out under suitable, dispersive conditions, it has been considered the Best Practicable Environmental Option. It is emphasised that the quantities of seafood processing waste are extremely low in comparison to the quantity of sewage sludge which was dumped in the past and the quantity of dredged material which currently requires disposal. Furthermore, it is emphasised that fishing activity removes a proportion of organic matter and shell material from the sea, which would otherwise contribute to organic matter and nutrient recycling and the formation of biogenic sediments. The disposal of waste products from this industry replaces only a small proportion of this material, which is naturally present in the environment (approximately 16% of the landings).

The report summarises the potential physical, chemical, biological and socio-economic impacts of waste disposal at sea. This highlights the cases where the addition of inert material, such as shell, can have a positive impact on the environment in terms of habitat creation/restoration, leading to increased species diversity, and mitigation against erosion in intertidal habitats. It also highlights cases where the addition of organic material would have a deleterious effect if the disposal site is chosen incorrectly. A summary of the scientific approach to detecting such impacts is also given, together with guidelines to selecting suitable sites for disposal.

The sea-disposal of fish and shellfish processing waste is concluded to be a viable option which can be accomplished in a sustainable way and which the six tenets of environmental management can be satisfied:

| Tenet of sustainable marine management: | Achieved: |
|---|---|
| Environmentally sustainable | as long as there is an adequate waste characterisation and site selection procedure |
| Technologically feasible | as long as the methods for suitable placement are devised |
| Economically viable | with economies of scale and a cost-benefit assessment |
| Socially desirable/tolerable | following agreement by stakeholders |
| Legislatively permissible | at a basic level but there is the need for clarification |
| Administratively achievable | as the statutory bodies and their advisors are in place |

That environmental sustainability is dependent on a satisfactory site selection and an appropriate site can be selected given a well-defined set of aims and the means to achieve those aims:

| Aim to: | Achieved by: |
|---|---|
| minimise interference with uses and users | desk-study and consultation |
| minimise the environmental impact | desk-study and fieldwork |
| evaluate options for disposal | desk-study and consultation |
| determine the capacity of the disposal site | desk-study, modelling and fieldwork |
| Characterise the receiving environment | desk-study and fieldwork |
| determine the transport of material thus influencing near and far field effects | desk-study and modelling with field validation |
| determine the accumulating/dispersing nature of the site | desk-study and fieldwork |
| consider the acceptability of any effects | desk-study and consultation |

As shown here, the sea-disposal of fish and shellfish waste requires a concerted assessment and further discussion between all parties. There is the need for a collaborative approach between the industry and the regulators with input from scientific, technical and economic expertise. This would indicate the way ahead to minimise or prevent problems. There is sufficient knowledge of marine processes and the assimilation of this type of waste to conclude that environmental impacts will not occur if the waste is disposed of in an appropriate manner. The use of the Best Practical Environmental Option procedure and by carrying out a sufficiently rigorous Environmental Impact Assessment will ensure that the above six tenets of sustainable environmental management are fulfilled and that all stakeholders are agreeable to the solution adopted.

# 1. INTRODUCTION

The disposal of seafood processing waste is a worldwide problem. For example, in Canada, the number of seafood processing plants, and hence, the amount of waste, has increased greatly since the 1960s (AMEC, 2003). Under the London Convention 1972 (formerly the London Dumping Convention, 1972), the dumping of fish offal is permitted and, in Canada, seafood processing waste is, wherever possible, sent to composting, fishmeal processing plants or, in the absence of suitable facilities, to landfill (AMEC, 2003). Shellfish waste is used in the production of lime, chitin and chitosan (Amec, 2003). However, in remote parts of Canada such as Newfoundland and Labrador, where waste cannot feasibly be sent for reprocessing or landfill, a number of permits have been issued allowing disposal at sea. As in the UK, the Canadian fish and shellfish processing industry is faced with an increasing problem of waste disposal and handling, particularly as regulations are becoming more stringent (Tidmarsh *et al.*, 1986; AMEC, 2003). Cost effective and environmentally sound solutions for waste handling and disposal are therefore required.

As in Canada, there has been large-scale increase in the seafood processing industry in Alaska. In the United States, the disposal of fish and shellfish waste at sea (in waters of >91 m depth) has, in the past, been permitted without a permit so long as the waste contained no additives (Champ *et al*., 1981). However, ocean disposal is considered by the USEPA (United States Environment Protection Agency) to be a last resort and is not considered to be the preferred method on the grounds of convenience or low cost.

Fish waste contributes significantly to organic waste generated by industry in Ireland but nuisance factors (such as odour) and legal limitations on disposal mean that the disposal of fish waste is now being recognised as a problem (Pfeiffer, 2003). There are a decreasing number of legitimate options for waste disposal from seafood processors, particularly as new licences for disposal at to landfill are no longer being issued due to EU regulations on the disposal of animal by-products (including fish). Therefore, Pfeiffer (2003) suggests that much illegal dumping currently takes place in Irish waters and the development of the seafood industry in Ireland may be constrained if viable waste management options cannot be found (Pfeiffer, 2003).

Within the UK, it is likely that Defra and the devolved administrations SEERAD and DARDNI will enforce an EU ban on sending untreated fish waste to landfill, despite the general lack of economically viable alternative waste disposal options (Johnston, 2004). Due to the recent tightening of regulations surrounding the disposal of seafood waste to landfill, fish and shellfish processors in the UK are now facing a rise in the cost of waste disposal as fish and shellfish processors will have to send their waste to a licensed renderer, composter or incinerator (Johnston, 2004). Therefore, alternative means of disposal, waste reduction and recycling, which are economically viable, environmentally sound and socially acceptable/tolerated, need to be found. Solutions to disposal must follow the six tenets for successful and sustainable environmental management (adapted from McLusky & Elliott, 2004):

- environmental sustainability;
- technological feasibility;
- economic viability;
- socially desirable/tolerable;

- legislatively permissible;
- administratively achievable.

The Sea Fish Industry Authority, as part of a review of the strategic framework for seafood waste management (Archer *et al.*, 2005), therefore wishes to investigate the potential for disposal of this waste at sea, the impacts of each particular type of waste at a disposal site and the means of choosing suitable disposal sites. This approach is required especially as the disposal of waste at sea is strictly regulated and requires careful consideration. The dumping of organic waste at inappropriate, especially at poorly dispersing sites, will cause problems with increased microbiological activity, the introduction of diseases and parasites and the production of anoxia in the sediments and bottom water in turn leading to the production of toxic substances such as hydrogen sulphide (OSPAR, 1998). This can have adverse effects on the ecology, especially the benthic invertebrates and fish, in the vicinity of the disposal site but also on the fishing industry itself in that fish populations may be affected, thus potentially reducing the catch. Therefore, as with the disposal at sea of any material, there will be the requirement to assess both near-field and far-field effects (MEMG, 2003).

## 1.1. Aim and Objectives

The present study aims to inform the debate regarding the feasibility of the sea disposal of fish and shellfish waste.

It therefore has the following objectives:

- to research the opportunities for disposal at sea and outline the current legislation surrounding the disposal of waste at sea;
- to define the types of waste and quantity of each type which require to be dumped at sea;
- to review the fate and ecological impacts of seafood processing waste disposal at sea, based on case studies where this has been carried out;
- to outline the scientific approach to impact detection and monitoring;
- to review the factors which need to be considered during the selection of disposal sites.

## 2. DISPOSAL AT SEA

In a telephone survey of UK processors, Large (2004) estimated that over 350,000 tonnes of waste was generated by the seafood processing industry in 2004, with 85% of this volume originating from finfish processing and the remainder arising from the processing of shellfish species. Whilst there may be several beneficial uses for this waste, most are not practicable (logistically) or economically viable for all parts of the UK (Archer, 2001). Therefore, the option of disposal at sea is currently being considered.

During the 1970s and 1980s, in the US, the disposal of seafood processing wastes at sea (both via pipelines and by dumping from boats) was regarded as the return of nutrients to the sea to support marine life and the process of recycling of products from the sea was similar to the natural process of death and decay. Thus, the return of heads, tails, viscera, blood and scales, provided that there were no chemical or biological additives, did not require a US EPA licence (Champ, 1981; Stevens & Haaga, 1992). However, in accreting areas this can lead to the accumulation of organic material which may result in a deterioration in sediment quality, low dissolved oxygen in the bottom waters and impoverishment of the benthic and epibenthic communities (Clarke, 2001).

The removal of fish and shellfish results in a reduction in the amount of organic matter and nutrients available for recycling in the sea although this may be a minor removal as most gutting of fishes takes place at sea. The total U.K. landed quantity for 2004 was reported as 457,713 tonnes (Defra statistics) of fish and shellfish. A survey by Large (2004) indicated a requirement to dispose of 71,452 tonnes at sea (16% of the reported landings. W. Lart, Sea Fish Industry Authority, pers. comm.). Since there was no differentiation between waste from imports and that from U.K. landed fish, there may be a proportion of this waste which originates from abroad. The amount of material returned to sea as a result of waste disposal is considerably lower than that taken out and, if carried out in an appropriate manner, should not cause the problems associated with the disposal of other organic wastes such as sewage sludge. Replacing some of the inorganic material (e.g. shell) may also enhance habitats, increasing or maintaining species diversity in areas which may have suffered habitat degradation as a result of fishing activities (Guay & Himmelman, 2004).

In Alaska, the Prince William Sound Science Centre (PWSSC) together with the Alaska Department of Fish and Game (ADF&G), the Alaska Department of Environmental Conservation (ADEC) and Cordova Seafood, have proposed a three year study (Cordova Fisheries Enhancement Project, 2004-2006) to determine the rate of decay and consumption of seafood waste dumped in the Northern Orca Inlet, Alaska (K. George, Alaska Department of Environmental Conservation, pers. comm.). In this study, the fish offal formerly considered as waste will be considered a useful by-product, providing a food source for aquatic life. That is, rather than dumping at sea as a means of waste disposal, time and effort will be taken to ensure that the material be placed in an area where it will be readily available for total consumption and assimilation over a relatively short period of time (1 month). As outlined in this proposal, the EPA (in 1992) stated that the dumping of waste at sea by barge was preferable to point source disposal from a pipe line as there was less potential for waste accumulation. The results of such studies will supplement the understanding of the current issues associated with disposal of seafood waste at sea, which are further discussed within this review.

## 2.1. Potential volume and composition of shellfish and finfish processing waste in the UK

The Seafish Waste Survey 2004 has enabled the present study to estimate the volume of shellfish and finfish waste that may require disposal at sea in the future (Large, 2004). For the purpose of this estimation, it was assumed that any present beneficial use of such waste would continue (e.g. fish meal production) and thus the potential amount of waste for sea disposal can be calculated from the amount of waste currently disposed of by non-beneficial uses such as landfill. The results of this survey can only be considered a guide and, in many areas, are likely to be an underestimate. If an application for a licence for disposal at sea were to be made, the maximum quantity per annum would need to be specified for a given site.

It was estimated that within the UK there is a potential for 45,238 tonnes per annum of shellfish waste (84% of the total shellfish waste) and 26,214 tonnes per annum of finfish waste (9% of total finfish waste) to be disposed of at sea (Table 1). This estimation can be taken further by separating the proportion of shellfish waste that originates from crustacean and mollusc species. In addition, the proportion of this waste made up of shell and organic matter (including offal) can also be estimated. In order to calculate these values, a number of further assumptions have been made:

- that the composition of shellfish catch comprised approximately 30% crustacea and 70% mollusca, based on the results of the Sea Fish Waste Survey 2001 (M. Archer, Sea Fish Industry Authority, pers. comm.);

- that crustacean processing waste was composed of 40% shell (exoskeleton) and 60% offal & organic matter (Pfeiffer, 2003);

- that mollusc processing waste was composed of 65% shell and 35% offal & organic matter (Pfeiffer, 2003).

Given these assumptions, it was calculated that, throughout the U.K., there is the potential requirement for the disposal of 5,429 tonnes of shell (exoskeleton) from crustacean processing and 20,583 tonnes of shell from mollusc processing (Table 2). This gives a total of 26,012 tonnes of shell waste that would require disposal by alternative methods such as disposal at sea. Similarly, there is the potential requirement for the alternative disposal of 8,143 tonnes of organic matter (including offal) from crustacean processing waste and 11,083 tonnes of organic matter (including offal) from mollusc processing waste (Table 2). This gives a total of 19,226 tonnes of organic matter and offal produced by the on-shore processing of shellfish species. However, these figures are only an approximate estimetion. Given the structural differences between the various harvested shellfish species, this analysis could be further developed at a species level if landing and processing information was available for the different species landed in each region.

There was no indication within the literature of the proportion of organic matter and inorganic waste products within finfish processing waste. However, it is expected that most of the 26,214 tonnes of waste comprises organic matter which may require alternative disposal methods, potentially at sea, in the future.

**Table 1. Potential amount of shellfish and finfish waste requiring disposal at sea (adapted from Large, 2004)**

| Region | Shellfish | | | Finfish | | |
|---|---|---|---|---|---|---|
| | Total waste (Tonnes per annum) | % NOT used for beneficial uses *** | Potential waste for sea disposal (Tonnes per annum) | Total waste (Tonnes per annum) | % NOT used for beneficial uses *** | Potential waste for sea disposal (Tonnes per annum) |
| South West England | 8,385 | 60 | 5,031 | 1,995 | 23 | 459 |
| Southern England * | 380 | 90 | 342 | 88 | 100 | 88 |
| Eastern England | 1,200 | 80 | 960 | 1,700 | 40 | 680 |
| Humberside | 250 | 100 | 250 | 177,225 | 1 | 1,772 |
| North East England | 815 | 80 | 652 | 1,360 | 40 | 544 |
| North West England | 3,200 | 55 | 1,760 | 3,200 | 10 | 320 |
| North East Scotland | 3,900 | 70 | 2,730 | 61,730 | 1 | 617 |
| Highlands & Islands | 3,730 | 75 | 2,798 | 38,625 | 55 | 21,244 |
| Central Scotland ** | 10,000 | 90 | 9,000 | 12,070 | 0 | 0 |
| South Western Scotland | 11,500 | 100 | 11,500 | 250 | 100 | 250 |
| Northern Ireland | 10,215 | 100 | 10,215 | 600 | 40 | 240 |
| Total | 53,575 | - | 45,238 | 298,843 | - | 26,214 |

* NOTE: Care should be taken with these results as they are based on a small sample
** NOTE: The shellfish figures should be treated with caution due to the small sample
*** NOTE: Potential waste calculated as worst-case scenario i.e. total waste not including that for beneficial uses.
100% was used where specific information was not present.

**Table 2. Composition of shellfish waste for potential disposal at sea (adapted from Large, 2004)**

| Region | Shellfish | | | | | | |
|---|---|---|---|---|---|---|---|
| | Potential waste for sea disposal (tonnes per annum) | Potential waste from Crustacea (tonnes per annum) | Potential waste from Mollusca (tonnes per annum) | Crustacean waste (Tonnes per annum) | | Molluscan waste (tonnes per annum ) | |
| | | | | Shell | Offal/Organic | Shell | Offal/Organic |
| South West England | 5,031 | 1,509 | 3,522 | 604 | 906 | 2,289 | 1,233 |
| Southern England * | 342 | 103 | 239 | 41 | 62 | 156 | 84 |
| Eastern England | 960 | 288 | 672 | 115 | 173 | 437 | 235 |
| Humberside | 250 | 75 | 175 | 30 | 45 | 114 | 61 |
| North East England | 652 | 196 | 456 | 78 | 117 | 297 | 160 |
| North West England | 1,760 | 528 | 1,232 | 211 | 317 | 801 | 431 |
| North East Scotland | 2,730 | 819 | 1,911 | 328 | 491 | 1,242 | 669 |
| Highlands & Islands | 2,798 | 839 | 1,958 | 336 | 504 | 1,273 | 685 |
| Central Scotland ** | 9,000 | 2,700 | 6,300 | 1,080 | 1,620 | 4,095 | 2,205 |
| South Western Scotland | 11,500 | 3,450 | 8,050 | 1,380 | 2,070 | 5,233 | 2,818 |
| Northern Ireland | 10,215 | 3,065 | 7,151 | 1,226 | 1,839 | 4,648 | 2,503 |
| Total | 45,238 | 13,571 | 31,667 | 5,429 | 8,143 | 20,583 | 11,083 |

* NOTE: Care should be taken with these results as they are based on a small sample

** NOTE: The shellfish figures should be treated with caution due to the small sample

NOTE: Assumed shellfish catch to be composed of 30% Crustacea and 70% Mollusca (M. Archer, Sea Fish Industry Authority, pers. comm.)

NOTE: Assumed Crustacea waste to be 60% offal & organic and 40% shell, and Mollusca waste to be 35% offal & organic and 65% shell (Pfeiffer, 2003)

For Northern Ireland, the figures for Crustacea probably an under estimate, the figures for mollusca are probably an over estimate (G. Griffiths, DARDNI, pers. comm.).

## 2.2. Legislation surrounding disposal at sea

### 2.2.1. Background

Since the end of 1998 the disposal at sea of most types of waste in the UK has been prohibited. The disposal at sea of radioactive waste ceased at the end of 1982, burning of waste at sea has not been permitted since 1992, dumping of industrial waste stopped at the end of 1992 and the dumping of sewage sludge ceased at the end of 1998 (Defra, 2004). Under international rules, fish waste from processing plants may be considered for disposal at sea although permissions will only be granted after careful assessment and under strict conditions established by the relevant licensing authorities or regulations (Defra, 2004). This section assesses specifically the international and national legislation present in the UK which controls the disposal of seafood waste at sea.

### 2.2.2. International Legislation

Two major international agreements, which therefore have required translation into EU and UK legislation, are of interest with respect to the disposal of seafood waste at sea: the 1992 OSPAR Convention and the 1972 London Convention.

The Convention for the Protection of the Marine Environment of the North-East Atlantic (the "OSPAR Convention") was opened for signature on 22 September 1992 and entered into force on 25 March 1998, replacing two previous conventions, namely the Oslo Convention and the Paris Convention (OSPAR Commission, 2004). Under the OSPAR Convention (Annex II, Article 3 paragraph 2) the dumping at sea of "fish waste from industrial fish processing operations" is permissible, although, such dumping must be authorised domestically under Article 4 of the Convention. In the UK, fish processing wastes may, subject to the other provisions of the Convention and the OSPAR fish waste guidelines, be disposed of at sea under FEPA II (see below).

The Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter, (the "London Dumping Convention", now the London Convention) was adopted on 13 November 1972 and entered into force on 30 August 1975 (London Convention, 2003). A new Protocol which was anticipated to replace the 1972 Convention was adopted on 7 November 1996 with the introduction of the "precautionary approach" to ocean dumping in Article 3. Under the 1996 Protocol, Article 4 states that Contracting Parties "shall prohibit the dumping of any wastes or other matter with the exception of those listed in Annex I". Annex I includes "fish waste or material resulting from industrial fish processing operations" and thus the disposal of seafood waste at sea is accepted under the Protocol. Furthermore, Article 9 requires all Contracting Parties to designate an appropriate authority or authorities to issue permits in accordance with the Protocol and so under the London Convention, fish processing waste may, subject to the other provisions of the Convention and the London Convention fish waste guidelines, be disposed of at sea under FEPA II licensing in the UK (see below).

### 2.2.3. National Legislation

In the UK, dumping of wastes at sea is prohibited, except under licences issued under Part II of the Food and Environment Protection Act 1985 – Deposits in the Sea (FEPA II) (Defra, 2004). The aim of FEPA II is to protect the marine environment from dumping of waste in tidal waters up to 200nm miles off the coast (i.e. within UK controlled waters). FEPA also applies to UK registered vessels anywhere in the world (Dr CMG Vivian, CEFAS, pers. comm.). FEPA II came into force on 1 January 1986 replacing the Dumping at Sea Act 1974, and has subsequently been amended by the following: the Environment Protection Act 1990; the Waste Management Licensing Regulations 1994; the Conservation (Natural Habitats &c) Regulations 1994; the Merchant Shipping (Consolidation) Act 1995; the Petroleum Act 1998; the Deposits in the Sea (Public Registers of Information) Regulations 1996; the Scotland Act; the Food Standards Act 1999; and the Countryside & Rights of Way Act 2000. It is of note that seafood waste discarded overboard immediately following capture is exempt from the FEPA II regulations (Dr C.M.G. Vivian, CEFAS, pers. comm.).

FEPA II provides the means for the UK to meet its obligations regarding the dumping of substances at sea, which is required under both the OSPAR Convention and the London Convention. In the UK, territorial responsibilities outside English waters, under FEPA, have been devolved to the Welsh and Northern Irish Assemblies and Scottish Parliament following the Government of Wales Act 1998, the Northern Ireland Act 1998 and the Scotland Act 1998, respectively (MCEU, 2005). The responsibility for the licensing of waste deposits in the sea (including seafood waste) fall within the remit of various departments and executive agencies within the various devolved UK countries. These structures are highlighted in Figure 1.

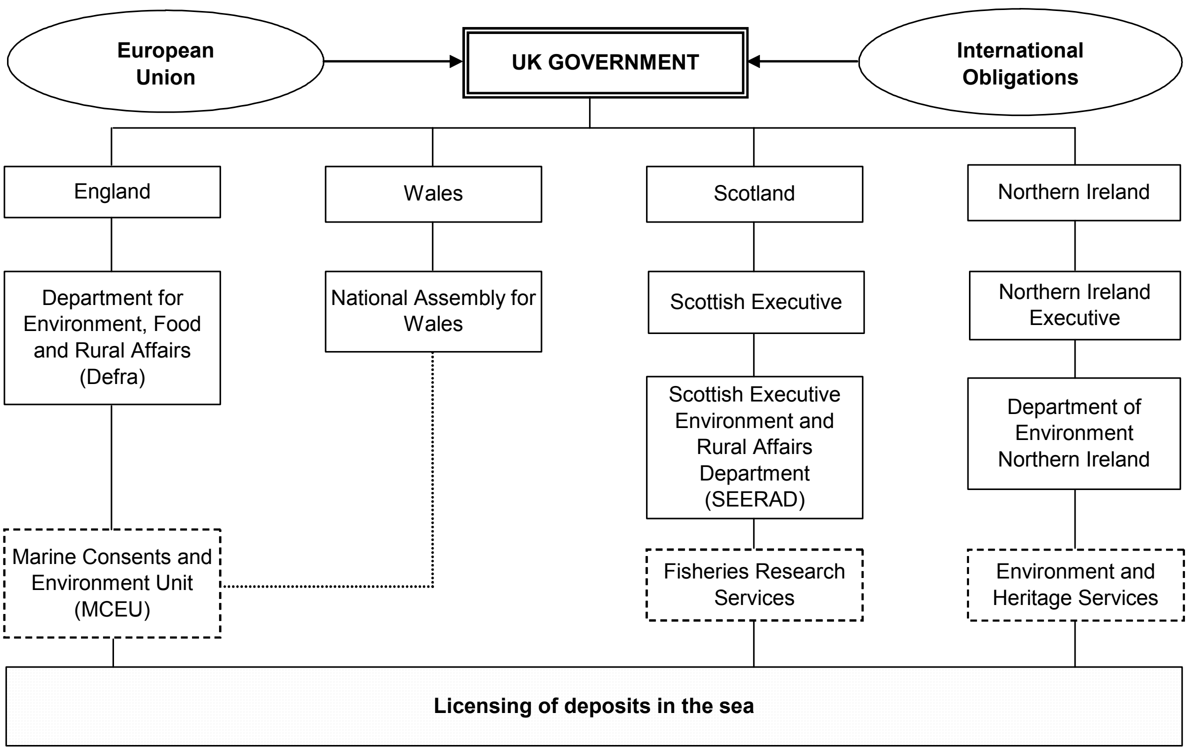


**Figure 1. UK Government. Responsible departments (solid boxes) and licensing agencies (dashed boxes) for deposits in the sea (adapted from Boyes *et al.*, 2003)**

In England, FEPA licences are issued by the Secretary of State for the Department for Environment, Food and Rural Affairs (Defra), and within Welsh waters licences are issued by the Welsh Assembly Government. The Marine Consents and Environment Unit (MCEU) administer the application of these licences on behalf of both of these licensing authorities and since 1 October 2004 Defra has been given full responsibility for the management of the Unit which was previously jointly managed by Defra and DfT (MCEU, 2005). The role of the MCEU is taken by Fisheries Research Services in Scotland (under the Scottish Executive) and the Environment and Heritage Services in Northern Ireland (under the Northern Ireland Executive). The licensing authority may grant licences for the dumping at sea with additional conditions attached; for example, dumping may only be undertaken at certain stages of the tide in order to reduce any potential impact. However, following land-based processing, it is of note that the dumping of fish processing waste at sea would only be considered by the above mentioned licensing authorities if it could be shown that other means of disposal are not allowed for sound ecological, social or economic reasons (Pfeiffer, 2003). Thus an assessment analogous to the Best Practicable Environmental Option (BPEO) procedure would be carried out. The legislative aspects of the disposal and utilisation of fish and fish processing waste in the Republic of Ireland are discussed in Pfeiffer (2003).

### 2.2.4. OTHER RELEVANT LEGISLATION

Under Section 34 of the Coast Protection Act 1949, the consent of the Secretary of State for Environment, Food and Rural Affairs is generally required for the “deposit of any object or materials below the level of mean high water springs” as the primary aim of the Act is to “ensure that works do not endanger the safety of navigation” (MCEU, 2005). The disposal of seafood processing waste at sea may therefore also require a permit to be issued under the Coastal Protection Act 1949, which has a 200nm limit following the introduction of the Continental Shelf Act in 1964 (MCEU, 2005).

The European Commission adopted the Animal By-products Regulation (EC 1774/2002) on 3rd October 2002, which establishes regulations for the collection, transport, storage, processing and use or disposal of all animal by-products. This legislation requires the categorisation of waste into three categories which are defined on their potential risk to animal or public health. Each category (1-3) has a defined range of permitted uses or disposal methods with most seafood waste (including shell and organic processing waste) being defined as Category 3 material (under Article 6 paragraph 1) and thus being regarded as little or no threat to the food chain provided they are treated effectively. However, the ABPR (defined above) does not currently permit disposal at sea. This aspect requires further discussion with U.K. regulators. Further information on the implementation of this Regulation is provided by the Defra website (Defra, 2005). For waste from specific locations, movement may be controlled under the E.U. Fish Health Regime based on Council Directive 91/67EEC and subsequent decisions. Further information can be obtained from the CEFAS website (www.cefas.co.uk).

# 3. CHARACTERISTICS OF SEAFOOD PROCESSING WASTE

Seafood waste arises from discards disposed of overboard during trawling, together with the processing (gutting, filleting, cooking, pickling, preserving and packing) of seafood products (Champ *et al.*, 1981). Seafood processing generates both liquid effluents and solid waste which contain both organic and inorganic materials. Liquid effluents can be screened to remove the settleable solids before being discharged but the effluent generally has a high BOD and a high oil, grease (applies predominantly to pelagic waste, not whitefish waste) and nitrogen concentration (due to the blood and slime) (Champ *et al.,* 1981; Amec, 2003). In general, the nitrogen and BOD concentrations are lower in effluents from shellfish processing plants than those from finfish processing plants (Amec, 2003).

Solid wastes can account for a very large proportion of the total waste for some species with solids comprising 30-60% of the total waste from filleting plants and 75-80% from crab processing (Amec, 2003). Of this, some of the waste will be organic (flesh) and some will be shell and bone. In general, seafood processing wastes will comprise processing waste from finfish, crustacean, and mollusc species. Finfish, particularly demersal species, may be purchased in numerous forms including whole, head-on gutted, headed and gutted, or fillet only (Archer, 2001). The resultant type and amount of waste will therefore be determined by the level of processing undertaken. Finfish waste will be comprised of entrails, heads, skeletal frames, skin, and bones in addition to organic matter still attached to these components after processing. This organic waste will consist of varying quantities (depending upon the species) of offal/viscera, skin, heads, tails and other solids, proteins, oil, grease, blood and slime (Champ *et al.*, 1981).

Molluscan waste is dominated by shell (65%) with the remaining waste comprising organic material which has remained attached to the shell after processing (Pfeiffer, 2003). The shell component will comprise whole shells and broken shells of varying size and shape depending on the species. It is also of note that approximately 7% of the weight of scallop waste (and presumably other filter feeders) is sand and silt that is washed out from within the scallops during processing/cleaning. This material may also require disposal at sea (W. Lart, Sea Fish Industry Authority, pers. comm.).

In contrast, crustacean waste is dominated by organic matter (60%) with the remaining waste comprising the discarded shell and carapace of crabs, lobsters and prawns (Pfeiffer, 2003). The composition of waste from, for example, *Nephrops* processing will be dependent on whether the *Nephrops* were originally landed whole or whether they were landed as shelled tails. It is also reported that for every kilogram of picked crabmeat, some 6kg of shell waste (including organic matter) remains which will have to be disposed of following processing (Leffler, 1997). Thus processing of different crustacean species will produce different volumes and types of waste.

All these types of waste behave very differently in the environment and the proportion and type of waste is an important consideration in whether or not disposal at sea is a viable option.

## 3.1. Fate of seafood waste

The fate of dumped seafood waste is controlled by a combination of factors including vessel practices and method of disposal, waste characteristics (buoyancy, size, shape etc), hydrodynamics of the system, and weather in addition to the environmental characteristics at the dumping site itself. The latter includes the biological, chemical and physical features of the site. All of these factors will contribute to the decomposition and dispersal of the waste once dumped in the marine environment. In general, the fate of seafood waste can be considered in three transitional phases: on the surface, in the water column and on the sea bed (Figure 2).

**On the surface**

Once the processing waste has been dumped at sea, the lighter, more buoyant fractions of the waste will float on the sea surface and, together with oil-based liquid waste, will form a surface slick. Blaber *et al.* (1995) found that discarded waste could remain on the surface for up to six hours. During this time the smaller fractions will be dispersed over the sea surface, whereas the larger/denser fractions will begin to sink.

The majority of the waste that remains on the surface is taken by scavenging sea birds and it was reported that in the North Sea between 1.4 and 3.4 million scavenging birds were known to feed on fishery waste during the winter (Camphuysen *et al.*, 1995 in Bluhm & Bechtel, 2003). Since birds tend to be size-selective in their feeding behaviour, this emphasises the importance of the size and weight of the waste discharges (Hill & Wassenberg, 1990 in Bluhm & Bechtel, 2003). The amount of seafood processing waste taken by sea birds may also be dependent on the time of day, weather and/or season (Bluhm & Bechtel, 2003) as these factors relate to the availability of other food sources and also the presence of migratory birds in the region.

In general any liquid waste that remains on the sea surface will be dispersed as a direct result of the local hydrodynamic and weather conditions at the chosen disposal site although oily waste will remain for a longer period than non-oily waste. Liquid effluents are diluted, dispersed throughout the water column and are generally carried away from the point of discharge by the currents. Hence, where dilution and dispersion is adequate, long term water quality problems should not arise.

**In the water column**

As the larger/denser material sinks towards the bed, it may be taken by mid-water scavengers such as pelagic fish and marine mammals although there is very little information regarding the amount of material taken in such a way. Stevens & Haaga (1992) reported that no fish were observed consuming particles of seafood processing waste in the water column during their study in Chiniak Bay, Alaska, although they did observe dense concentrations of euphausids among the particles within several metres of the sea bed. Hence, the proportion of material taken by mid water scavengers may be of minor importance in comparison to that taken by sea floor scavengers (Bluhm & Bechtel, 2003).

**On the seabed**

Following dumping, much of the waste sinks directly to the bed, the rate being dependent upon the fat, muscle/flesh and shell content and the presence or absence of an intact swimbladder (where fish will float if the gas is not expelled) (Bluhm & Bechtel, 2003). It was noted by Stevens & Haaga (1992) that heavier parts of discarded seafood waste (including heads and whole fish) sank quickly, medium sized pieces (2-25cm) of gills, skin, viscera, fins etc required about 0.5 hr to reach the sea bed whereas the smallest particles required more than an hour to reach the sea bed 150m below the surface in Chiniak Bay, Alaska.

In low energy areas, the waste accumulates on the sea bed covering only a small area, but with the waste forming relatively deep mounds. In higher energy areas, the waste is dispersed over a larger area, with relatively shallow layers of waste accumulating on the sea floor. Once on the sea floor, organic material has an oxygen demand which, in areas of low disturbance, will produce anoxic waters and underlying sediments. This leads to changes in the bacterial, macrofaunal and scavenger populations. These effects are discussed in more detail in Section 4.

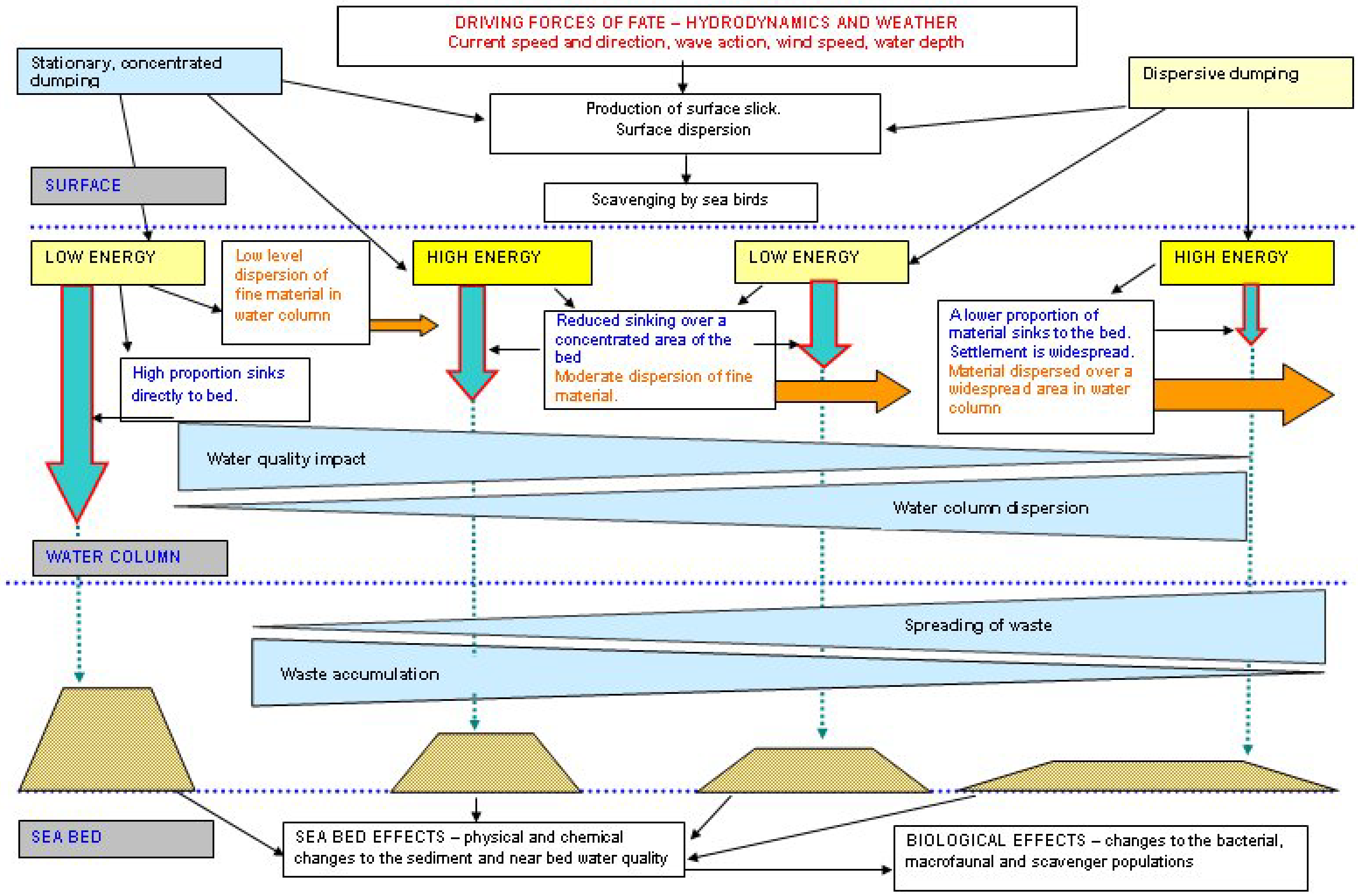


**Figure 2. Fate of seafood waste**

# 4. ECOLOGICAL AND ENVIRONMENTAL IMPACTS

Amec (2003) did not locate any reports on the effects of the dumping of seafood processing wastes at sea. Stevens & Haaga (1992), Bluhm & Bechtel (2003) and K. George (Alaska Department of Environmental Conservation, pers. comm.) also highlighted the lack of information surrounding this subject although some recent studies have examined the effects of the discharge of liquid wastes generated by the seafood processing industry (e.g. Rudolph *et al.*, 2002; Ramøn *et al.*, 2004). Similarly, several monitoring and impact studies have been carried out around fish and shellfish farms.

Aquaculture waste, and that generated from seafood processing plants, contains varying quantities of nutrients, ammonia and both organic (viscera) and inorganic (e.g. shell and bone) particulate matter (NEFMC, 1998) and so some of the effects of this would be similar to those associated with the dumping of solid organic waste. The effects of fish farms, and the rate of organic matter deposition therefore need to be examined as a model when considering the disposal of seafood processing waste at sea. However, it should be noted that fish farms are generally situated in low energy areas, close to the shore, and the deposition of organic matter occurs in a highly concentrated area (McLusky & Elliott, 2004). It should also be noted that much of the waste from fish farms consists of uneaten food pellets and faeces which are rich in nutrients and organic matter. These components are not present in the waste generated by the seafood processing industry.

The ecological impacts of the disposal of the organic component of the solid waste from the seafood processing industry are also likely to be similar to those associated with the dumping of sewage sludge and the disposal of organically-enriched dredged material at sea. As stated above, the effects of seafood waste disposal at sea have not been well studied. The following section therefore gives a generalised account of the effects that would be expected as a result of the disposal of organic waste.

The characteristics of seafood processing waste, and its fate following disposal at sea are described in Sections 3 and 3.1. The two major classes of waste (in terms of their potential environmental effects) are predominantly organic (which may contain some shell and bone material) and predominantly inorganic shell (which may contain varying amounts of organic tissue). The environmental effects of each type seafood processing waste are summarised in Figures 3 - 6 (organic/mixed component) and Figure 7 (shell component). The magnitude of the effect has two components – the spatial extent and the duration (temporal aspect) of the effect. These effects are largely influenced by the hydrodynamic regime, the volume, particle size and buoyancy of the waste and its composition in terms of the proportion of organic and inorganic material. These factors govern the rate of sinking, degree of dispersal in the water column and the degree of dispersal or accumulation of the waste on the sea bed, its availability to species which may utilise it as a food source and the overall rate of degradation and removal from the marine environment.

The dumping of such material will affect the physical and chemical characteristics of the seabed which will, in turn, impact upon the benthic communities and eventually, species at higher trophic levels (Elliott *et al.*, 1998). The accumulation of waste piles or infilling of channels (such as those created by the residual current patterns) may result in an alteration of the hydrodynamic regime, which is also linked to the ecological functioning of the area and the benthic and pelagic communities inhabiting it (Elliott *et al.,* 1998).

## 4.1. Effects of organic waste

### 4.1.1. WATER COLUMN EFFECTS

Ramøn *et al.* (2004) highlighted the environmental problems faced by Chile and Peru as a result of the discharge of liquid seafood processing waste. The main pollutants in this waste included large amounts of organic matter, oil and grease residues, the decomposition of which cause suboxic and, in severe cases, anoxic conditions in the water column and the sediment (Chareonpanich *et al.*, 1994; Tsutsumi, 1995; Boyra *et al.,* 2004). Increased turbidity caused by suspended solids in the effluent together with the emulsification of grease and oils reduced light penetration, and thus had the potential to reduce photosynthesis and direct oxygen exchange at the air-water interface (Elliott *et al.*, 1998; Ramøn *et al.*, 2004).

In Lota Bay, Chile, such effects were recorded at their maximum in areas where the highest discharge of organic material (from fish processing) occurred. These effects were found to be seasonal and coincided with high temperatures and maximum fishing (and hence fish processing) activity (Ramøn *et al.*, 2004). Rudolph *et al*. (2002) studied the impact of the discharge of liquid effluent from fish processing plants on San Vicente Bay, Chile and found a seasonal pattern of oxygen depletion with a 95% oxygen deficit (5% saturation) during spring. This was considered to be related to the high concentration of organic matter in the water column and on the sea bed which had originated from the discharge of industrial, including approximately 63 kg $d^{-1}$ of seafood processing, waste. Carrasco (1996, in Rudolph *et al*., 2002) stated that this problem had been observed in most of Chile's bays, particularly those receiving fisheries waste. However, the long residence time of water in San Vicente Bay would exacerbate the problem. Several authors have also reported oxygen depletion in the bottom water, anoxic conditions in the sediments and subsequent impacts on the benthic communities in areas influenced by fish farms (Chareonpanich *et al.*, 1994; Tsutsumi, 1995; Boyra *et al.,* 2004).

### 4.1.2. SEA BED EFFECTS

As stated above, the increased biochemical oxygen demand, resulting from the microbial degradation of organic matter, can reduce the oxygen concentration of both the sediments and the overlying water, and lead to hypoxia or anoxia. The small particle sizes which can be associated with organic wastes can further inhibit oxygen penetration to the sediment, by blocking interstitial spaces, and the deposition of both large and small particle sizes can lead to smothering of the bed if the currents are insufficiently strong to prevent accumulation (Elliott *et al*., 1998; Belias *et al*, 2003).

The effects of organic pollution on the marine and estuarine environment are well documented and were reviewed, predominantly in relation to discharges from sewage treatment works and pulp mills, by Pearson & Rosenberg (1978) and Diaz & Rosenberg (1995). Furthermore, models have been developed (e.g., BenOss – Biological effects and Organic solids settlement (Cromey *et al*., 1998); Henderson *et al*., 2001) to enable the prediction of carbon accumulation on the sea bed and the associated effects on the benthic community. A high concentration of organic material provides a substrate for large bacterial populations, thus giving a high Biochemical Oxygen Demand (BOD) (Rhoads, 1974). Bacterial mats, composed of *Beggiatoa* spp (sulphur oxidising bacteria) have commonly been identified in areas receiving large amounts of organic material be it from fish farms (Rajendran *et al*., 1999; Christensen *et al*., 2000; McLusky & Elliott, 2004), sewage sludge-disposal (e.g. Garroch Head, MacKay, 1986) or the disposal of seafood processing waste (Stevens & Haaga, 1992; Tetra Tech, 2004). A study beneath fish cages in the Philippines showed sediment metabolism to decrease with increasing sedimentation suggesting that microbial activity had reached saturation and that anaerobic decomposition processes were dominating (Holmer *et al*., 2003) (although the high levels of faeces and food pellets present in fish farm waste are not present in seafood waste). Tsutsumi (1995) stated that the continual addition of organic matter can exceed the decomposition capacity, i.e. the capacity of the receiving environment to degrade, disperse and assimilate the waste. This problem can be exacerbated in areas of fine sediments or where the material being disposed of contains a large proportion of fine particles which block interstitial spaces, reducing permeability and oxygen penetration (Rhoads, 1974; Elliott *et al*., 1998). Furthermore, high summer temperatures lead to increased microbial activity and reduced oxygen solubility, increasing the potential for the development of anaerobic conditions.

Anoxia in the interstitial (pore) waters of the sediments can eventually lead to the formation of methane, ammonia/ammonium ions and hydrogen sulphide (Pearson & Rosenberg, 1978; Libes, 1992; Belias *et al.,*2003; Holmer *et al*., 2003). The decay process of fish and shellfish also releases significant amounts of ammonia and nitrate (Leffler, 1997). During dive surveys carried out by Tetra Tech (2004), gas bubbles (presumably methane) were seen escaping from piles of discarded fish heads at a seafood processing waste disposal site near Kethikan, Alaska. Such observations were also made at other sites by Belias *et al.* (2003) and Holmer *et al.* (2003).

Monitoring throughout the UK has shown that the primary effect of the disposal of sewage at sea has been organic enrichment, the most notable case being the Garroch Head site in the Firth of Clyde (Heap *et al.,* 1991; MAFF, 1993). However, secondary, toxic effects have been noted in some cases as a result of the concentration of trace metals and other pollutants which may be present in sewage sludge (MAFF, 1993). Whilst the adverse effects of organic enrichment were not detectable at the Barrow Deep site, a toxic effect, leading to reduced species diversity and numbers, was recorded at a few sampling stations. Such toxic effects arise as a result of the liberation of contaminants upon exposure to some form of chemical change. For example, changes in the aerobic/anaerobic balance in the sediments may affect the chemical state of the contaminants in question, their diagenesis within the sediments, their affinity for binding to particulate matter or their solubility (Calmano *et al.,* 1996).

Similarly, fish and shellfish are known to accumulate potentially toxic substances such as metals and organic compounds (e.g. PCBs and dioxins). Whilst these levels may be low at an individual scale, the disposal of mildly contaminated waste could lead to a concentration of toxins which, depending upon the seabed conditions, could cause liberation to the water column making elevated concentrations of these substances available to other organisms. Archer (2004) provides a summary of the concentrations of various metals and PCBs in fish and shellfish species although, given that processed seafood is considered fit for human consumption, concentrations of such substances are low. Levels of naturally occurring toxins, such as those causing diarrhetic shellfish poisoning (DSP) and amnesic shellfish poisoning (ASP), are currently monitored and shellfish harvesting is prevented if levels are high. Therefore, the potential for the release of toxins at concentrations high enough to cause water quality problems is low on a routine basis but should not be overlooked.

### 4.1.3. FAUNAL COMMUNITIES

The accumulation of organic matter on the sea bed will adversely affect the faunal community. Low dissolved oxygen in the water column, together with the increased turbidity caused by particulate matter in the waste, can reduce water quality to the extent that mobile species (fish, crustaceans, mammals) either avoid the area or simply cannot survive leading, to an overall change in the community structure – that is, a temporary water quality barrier is created (MAFF, 1993; Elliott & Cutts, 2004). Dimech *et al*. (2002) reported significant differences in the diversity of echinoderms, decapods and molluscs between unimpacted sites and those influenced by caged fish farms. Kakuta & Murachi (1997) reported a number of physiological responses in carp (*Cyprinus carpio*) exposed to sewage, together with abnormalities in the kidney, pathological effects to the gills and mortality in all fish exposed to raw sewage containing fish processing waste. Lawrence & Hemingway (2003) review the sublethal and lethal effects of pollutants in fishes.

Similarly, such conditions together with the presence of toxic substances such as hydrogen sulphide and ammonia can lead to an impoverishment of the benthic community. Communities associated with organically enriched or polluted environments are commonly composed of very few species, present in high abundances with small body size and low community biomass (Figure 3). The organisms present are usually opportunistic (*r*-strategists) with growth and reproduction characteristics which allow them to take immediate advantage of sudden environmental changes providing them with a suitable habitat. These species are tolerant, at least for a sufficient length of time to reproduce, of the conditions associated with organic enrichment or pollution (Pearson & Rosenberg, 1978; Gray, 1982; Pearson *et al*., 1982; Gray *et al*., 1988; Yokoyama, 2002; McLusky & Elliott, 2004).

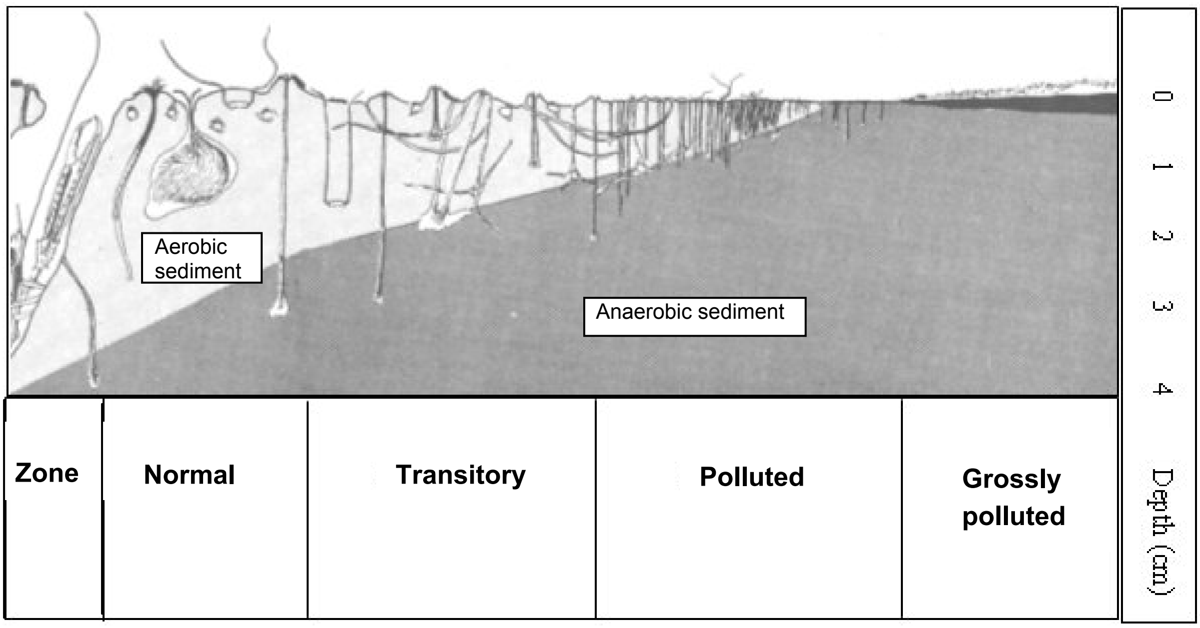


**Figure 3. Typical changes in fauna and sediment structure along a gradient of organic enrichment (redrawn from Pearson & Rosenberg, 1978).**

Pearson & Rosenberg (1978) used these basic, quantitative, parameters to derive the species, abundance, biomass (SAB) curves (Figure 4) which have been widely used as an indication of organic pollution impacts on marine benthic communities. Sediments at the point of maximum pollution are generally devoid of life but as the organic input decreases slightly, the sediments become colonised by large numbers of small opportunistic organisms. Biomass and the number of species remain low but at the peak of opportunists (PO), there is a temporary peak in biomass which is due to the shear number of organisms. As the organic input further decreases, the number of species increases whilst abundance and biomass both decrease. This area of low biomass, caused by the dramatic decrease in animal abundance, is known as the ecotone (E) and marks the point where two distinct community types merge. Throughout the transition zone (TR), as organic input further declines, there is a progressive change from a community characteristic of polluted sediments, through a community which benefits from slight organic enrichment, to one which is characteristic of undisturbed conditions (Figure 4). Species commonly found in organically enriched areas include capitellid polychaetes (*Capitella capitata, Malacoceros fuliginosa)* nematodes and oligochaete worms (Pearson & Rosenberg, 1978; Chateonpanich *et al*., 1994; Mirto *et al*., 2002).

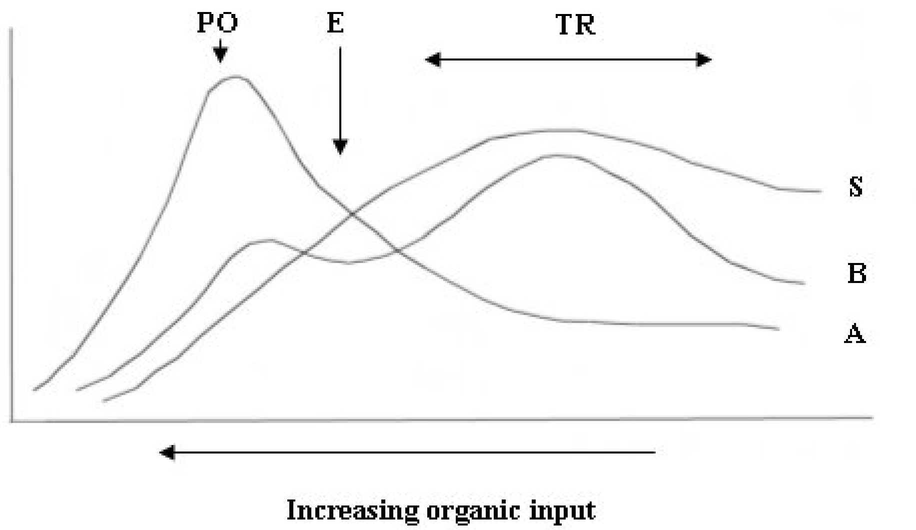


**Figure 4. Generalised SAB diagram (re-drawn from Pearson & Rosenberg, 1978) showing changes in species (S), abundance (A) and biomass (B) along an organic gradient. PO = Peak of opportunists; E = ecotone point; TR = transition zone.**

Tsutsumi (1995) noted that oxygen depletion and reducing conditions in the sediments occurred during the summer, sometimes leading to defaunation whilst recolonisation took place during the autumn and winter. Following organic enrichment of the sediments, the benthic community switched from one dominated by bivalves to one dominated polychaetes. Eden *et al*. (2003) observed a seasonal pattern in the abundance of the mud snail *Nassarius sinusigerus* with the highest numbers being recorded in areas showing moderate impacts from the effects of a fish farm. During the summer, the degree of impact in the vicinity of the fish farm increased so that the area of moderate impact, together with the maximum abundance of snails, was created away from the farm. As conditions improved during the cooler months, the maximum abundance of snails moved back towards the fish farm. The distribution of this species around fish farms was found to be determined by the balance between the attraction of the snail to organically enriched sediments (hence increased food supply) and deterrence due to anoxic, sulphide sediments resulting from over enrichment. Blanchard & Feder (2003) noted a marked increase in opportunistic fauna (particularly cirratulid and capitellid polychaetes) following the accumulation of fish processing waste (from an outfall) in the Port of Valdez, Alaska.

Monitoring carried out around a number of offshore sewage sludge disposal sites showed the Garroch Head site in the Clyde Sea to be organically enriched as a result of disposal activities (MAFF, 1993). That is, the benthic communities showed a typical Pearson-Rosenberg response along an organic gradient with the communities in the centre of the dump site being dominated by one or two species, including *Capitella capitata,* inhabiting highly anoxic sediments. With increasing distance from the site, and decreasing organic content, an increase in species diversity and a reduction in organism abundance were recorded, together with increasing oxygenation of the sediment. The faunal community became more representative of the area, approaching a climax community characterised by high diversity and evenness (i.e. no overall dominance by one or few species), lower abundance and larger, deep burrowing species characteristic or aerobic sediments (Figures 3 and 4).

However, the Garroch Head site was situated in a low energy (quiescent) muddy area where settlement and accumulation of waste was high and dispersion was low. Therefore, the receiving environment became degraded. At other, more dispersing, sites around the UK, such responses were not observed and any impoverishment of the benthic community which may have been recorded could not be attributed to the disposal of sewage sludge. For example, the Barrow Deep (outer Thames estuary) disposal site was situated in a high energy, dispersive area where the benthic community was naturally impoverished (maintained at an early successional stage) as a result of strong, natural environmental fluctuations (e.g., tidal currents and wave action). The low numbers of species, high dominance and erratic fluctuations in the abundance of certain species recorded were considered typical of such an area. The bed in this area was composed of sand and muddy sand in a series of channel systems. In the lower energy areas, some settlement and accumulation of organic matter was noted which led to localised increases in species diversity and abundance. This enhancement of the community as a result of the dumping of sewage sludge was considered to be beneficial (MAFF, 1993) and did not represent the effects of over enrichment associated with impoverished communities. Similarly, monitoring at the Tyne disposal site revealed only mild evidence of organic enrichment whilst no effects of sewage sludge disposal could be detected at the St Abbs site (Heap *et al.,* 1991; MAFF, 1993). Therefore, the distinction between natural and anthropogenic controls over the benthic community structure and sediment properties could not be made.

As indicated above, whilst a classic response of the benthic community to organic enrichment was observed at Garroch Head (a low energy, quiescent area promoting settling), it is of note that in several other, higher energy, dispersive areas, there was very little evidence of any significant change in the benthic community resulting from the dumping of sewage sludge. MAFF (1993) therefore, concluded disposal at sea to be a viable and environmentally acceptable option. This conclusion, of course, related to disposal in high energy areas. It should be noted that up until 1998, when the disposal of sewage sludge at sea ceased, between 8 and 10 million tonnes of sewage were dumped annually around the

UK (MEMG, 2003). By comparison, this review is considering the disposal of 71,000 (approximately) tonnes per annum of waste, of which approximately 20-25,000 tonnes is inorganic shell material with 45-50,000 tonnes of organic material (Tables 1 and 2). Given the conclusion of MAFF (1993), it is not anticipated that the disposal of this amount of waste will cause adverse environmental effects provided that disposal takes place in a dispersive area and that the waste is dumped in small amounts and spread to minimise the potential for accumulation.

Similarly, adverse environmental impacts have not been reported at all fish farm sites. For example, Merceron *et. al.* (2002) monitored water quality around a marine fish farm, in a well flushed area, and did not record any impact on dissolved oxygen, suspended particulate matter, phosphate or nitrate levels. Ammonia concentrations were slightly elevated in the immediate vicinity of the cages but this impact was extremely localised. Chamberlain *et al.* (2001) also demonstrated the lack of impact of large mussel farms on the sea bed in low energy areas but where the current patterns were such that accumulation of organic material did not occur. Karakassis *et al*. (1999; 2000) indicated that the impacts of fish farming (or the deposition of organic matter from other sources) on the benthos, and the recovery of the benthos following the cessation of aquaculture, were variable according to the site characteristics.

With regard to the effects of seafood processing waste, small scale dumping was licensed by FRS (Fisheries Research Services, Aberdeen) at a site close to the Orkney Islands until c.2002. Little monitoring was carried out at this site but the seabed showed no visible signs of impact (J. McKie, FRS Aberdeen, pers. comm.). Waste from salmon processing has also been discharged into Loch Creran where, in the past, there has been concern over oily slicks, ammonia and BOD which caused anoxia and discolouration of the water (P. Holmes, Scottish Environment Protection Agency, pers. comm.). This waste is now treated and SEPA do not believe that there are any impacts on the sea bed.

#### 4.1.4. PRIMARY PRODUCTION

Primary production and phytoplankton growth can also be reduced in the water column as a result of increased turbidity caused by the input of suspended solids and the emulsification of grease and oils (Ramøn *et al.*, 2004). However, increased algal growth may become problematic when nutrient enrichment leads to eutrophication and, in some cases, harmful algal blooms which may lead to toxic shellfish poisoning (NEFMC, 1998; Amec, 2003). Boyra *et al.* (2004) also found increased growth of pollution tolerant macroalgae, as a result of nutrient enrichment in areas influenced by fish farming, together with an increase in detritivores and filter feeders in the intertidal zone. Conversely, smothering of macroalgae and seagrass beds in shallow areas may also occur (Dimech *et al.*, 2002; Cancemi *et al.*, 2003).

Changes to the sediment conditions through an increase in fine particles together with increased growth of epiphytes resulting from nutrient enrichment led to reduced plant growth and plant mortality in Mediterranean *Posidonia oceanica* meadows in areas around fish farms (Dimech *et al.*, 2002; Cancemi *et al.*, 2003). Ruiz *et al.* (2001) reported a loss of 11.3 out of 39 ha of *P. oceanica* meadow with a further 9.9 ha being degraded since the onset of fish farming in south eastern Spain. Whilst *P. oceanica* meadows do not exist in UK waters, there are a number of *Zostera noltii/marina* beds around the coast which are protected under the Habitats Directive (92/43/EEC) (Elliott *et al.*, 1998) and, as demonstrated by Boyra *et al*. (2004), the impacts of organic waste (in this case, from fish farming) are not restricted to subtidal areas.

#### 4.1.5. OTHER EFFECTS

Other effects of organic wastes include direct toxicity through exposure to ammonia, methane and hydrogen sulphide generated during the decomposition of organic matter. All marine species are susceptible to, and most are infected to a certain degree by, parasites and disease. Therefore, the dumping of seafood processing waste at sea gives rise to the potential for the movement of disease and parasites between habitats (OSPAR, 1998). Movement of certain wastes may be governed under the EU Fish Health regime to prevent the spread of specific notifiable diseases (see section 2.2.4.). Although disease and parasites are naturally present and are dispersed around the marine environment as a result of the movement of infected mobile species, the storage of the waste and thus the concentration of disease organisms, pests or parasites can lead to high levels of them in waste holding facilities. With hygienic onshore storage facilities, the waste should not become hazardous in U.K. practice (Dr CMG Vivian, CEFAS, pers. comm.).

As will apply to all aspects of disposal at sea, the concentration of waste in one area could have adverse impacts including increased infection rates of the organisms in the receiving area (OSPAR, 1998). For example, the disposal of seafood processing waste has been associated with winter mortalities of sea otters which were found to feed on waste material when food was limited. These deaths were attributed to infection by a helminth parasite (*Pseudoterranova decipiens*) whose intermediate host includes fish (Cordova Fisheries Enhancement Project Proposal, K. George, Alaska Department of Environmental Conservation, pers. comm.). Sindermann (1979) also reported gill necrosis in crustaceans as a characteristic effect of exposure to organic pollution. There are also diseases which affect wild populations of fish and shellfish such as the dinoflagellete *Hematodinium* which is present in *Nephrops* (Briggs & McAliskey, 2002; Stentiford & Neil, 2004) and other crustaceans (Stentiford *et al.*, 2003). The extent of these infections is only recently becoming known (D. Neil, University of Glasgow, pers. comm.). A more cautious approach to the disposal of imported shellfish is also required since, for example, lobsters imported from North America may be a source of *Gaffkemia* (Lavallee *et al*., 2002). The risk of transfer of disease via disposal of seafood waste from imported fish and shellfish should be taken seriously since it could potentially result in the introduction of non-indigenous diseases to U.K. fish and shellfish stocks.

As stated above, infection is natural but the concentration of infected fish can lead to the infection of larger numbers of predators than if the source of the parasite was widely dispersed. However, if dumped over a large, highly dispersive area, the spread of infection should not be any greater than is natural.

Champ *et al.* (1981) stated that slicks associated with the offshore dumping of seafood processing wastes in the United States during the 1970s, has led to a number of oil spill alerts due to the surface film caused by fish oils. Tidmarsh *et al.* (1986) reported offal on beaches, fouled nets and detrimental effects on lobster fisheries in the vicinity of disposal sites off the coast of Newfoundland. Poor circulation in Fogo Harbour, Newfoundland, led to the development of an 1800 $m^2$ abiotic area following the dumping of seafood processing waste. However, it was suggested that such effects could be minimised by dumping at a site with stronger residual currents and larger scavenger populations, thus utilising the assimilative capacity of the receiving environment.

With reference to dredged material, disposal at sea is, in many cases, considered to be both the best practicable environmental option (BPEO) and the most economically viable one, if carried out in an appropriate manner (MEMG, 2003). However, the dumping of seafood processing waste at an inappropriate site may have adverse effects of the seafood industry, for example, poor water quality may adversely affect catches. The effects of dumping seafood waste at sea are straightforward on a conceptual basis but difficult to predict on a quantitative basis. However, these effects may be minimised by the careful choice of site and the determination and use of the receiving area's assimilative capacity. Despite this, given the current adoption of the Precautionary Principle by the contracting parties to the OSPAR and London Conventions, the disposal of the waste will not be permitted unless it can be shown to produce no adverse effects. In addition to considering the assimilative capacity of the receiving environment, there is the need to quantify the appropriate discharge rate and monitoring following disposal in order to detect ecosystem change before it becomes irreversible (Champ *et al.*, 1981).

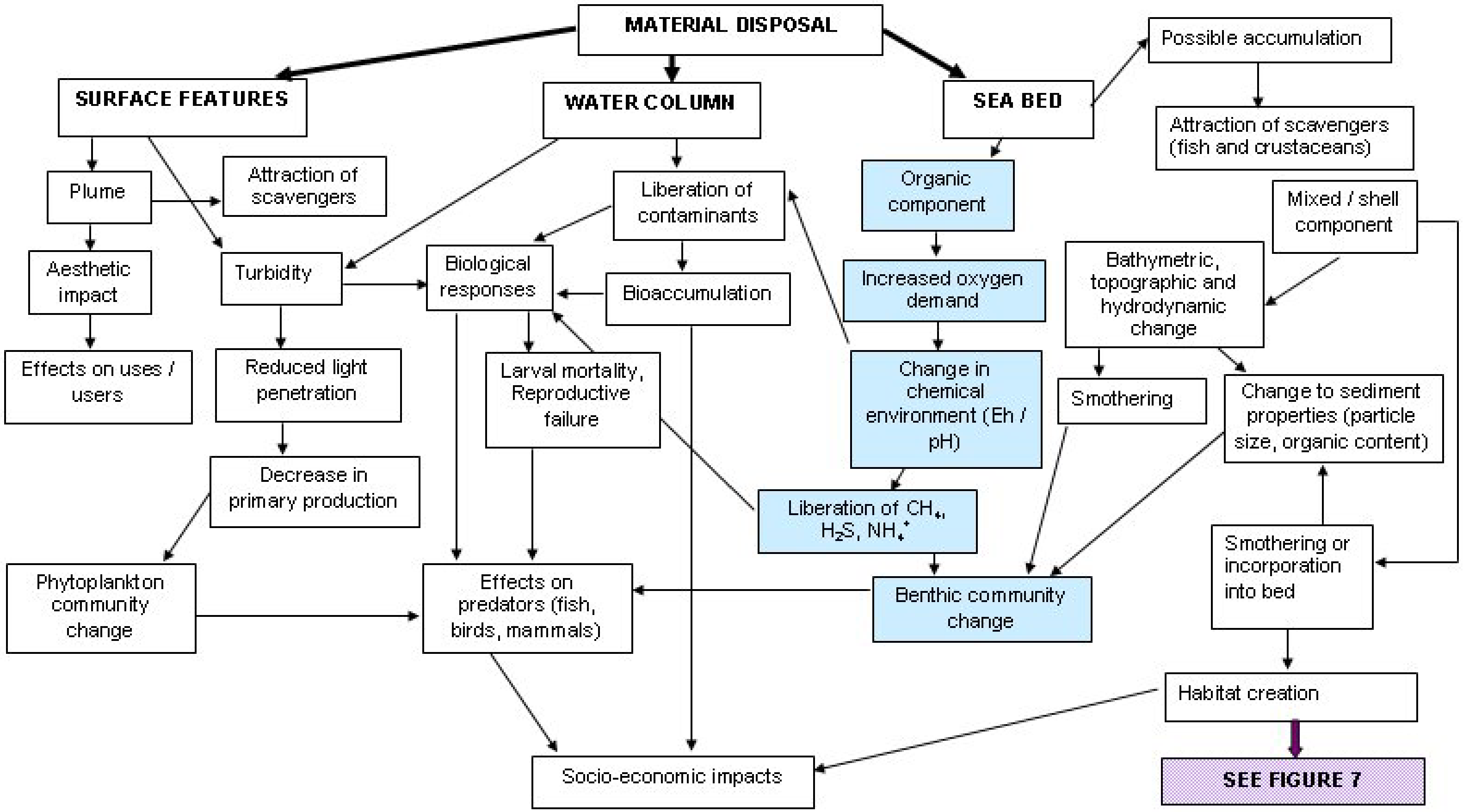


**Figure 5. Potential environmental impacts of marine disposal of fish processing waste – conceptual model. (A more detailed description of the processes in the blue boxes is given in Figure 6).**

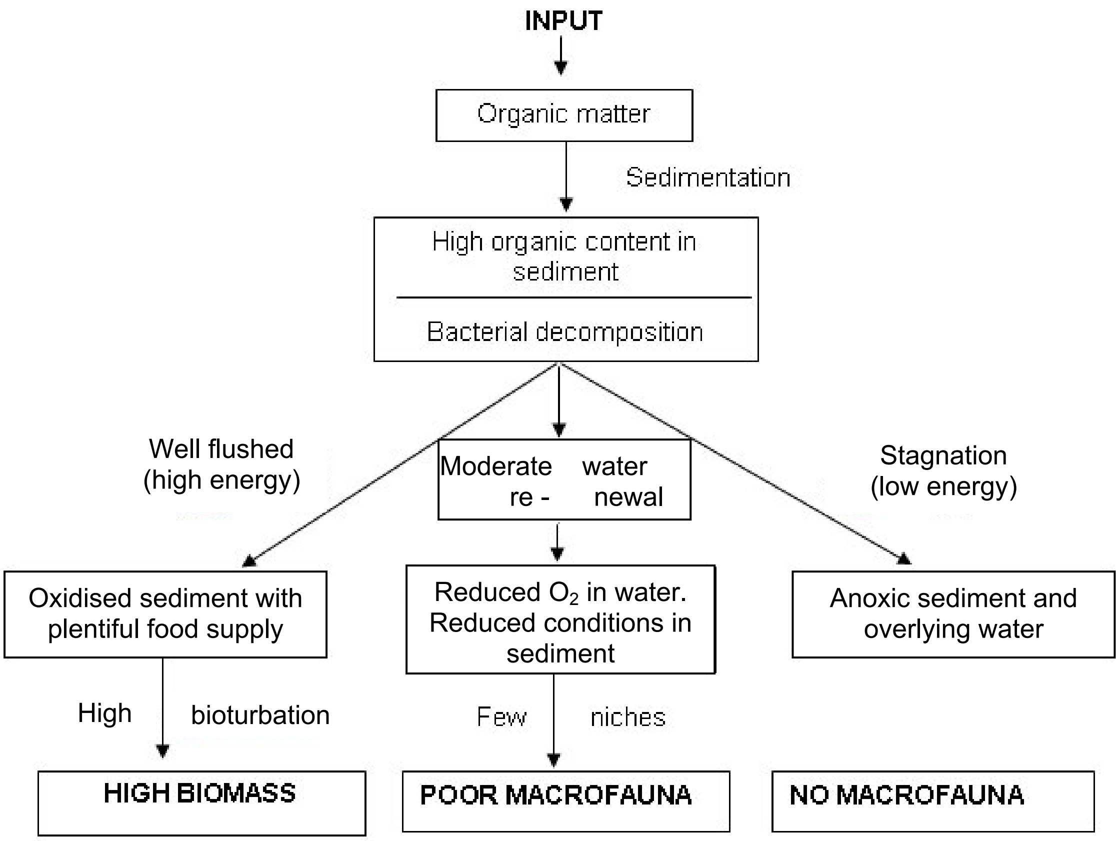


**Figure 6. Some of the pathways of organic material in the marine environment and its effects in relation to water renewal (from Pearson & Rosenberg, 1978).** (Bioturbation refers to sediment mixing by infaunal organisms which can enhance organic matter decomposition and other sediment processes).

## 4.2. Effects of shell disposal

Very little information is available on the fate of discarded shell material in the sea (be it natural or anthropogenic). However, it is assumed that, provided it is cleaned of most of the organic material, shell fragments will be relatively inert and are therefore will not adversely affect water quality. It is anticipated that if the material is dumped in a high energy area, at which the sea bed is composed of coarse sand, gravel and shell material, then the shell waste will naturally disperse and become incorporated into the existing sediment. This will rely on the material being within a particle size range characteristic of the receiving sediment which can easily be determined by sampling the sediments prior to disposal. In addition, the material must be dumped in appropriate quantities so that dispersal rather than accumulation occurs.

Dr K Collins (Southampton Oceanography Centre, pers. comm.) stated that in the English Channel (east of the Isle of Wight), aggregate extraction results in the removal of the surface shell layer, leaving a purely sand gravel seabed. The biogenic shell component is thought to provide a key settlement habitat for several species and it was proposed that replacing this shell material may enable rapid restoration of the seabed and the associated benthic communities. Whilst studies need to be carried out to confirm this, it may provide a viable, inexpensive and environmentally beneficial means of disposal of the shell component of the waste generated by the seafood processing industry. Similarly, the disposal of certain types of dredged material has been found to have a positive impact on benthic communities, fish populations and plant species (Beynon *et al.*, 2000). However, whilst the removal of large amounts of shell material from the sea bed during aggregate extraction can have adverse effects on the benthic community, this is thought to be a minor consideration since aggregate dredging areas do not have significant levels of shell as its presence in aggregates affects the quality and strength of concrete (Dr CMG Vivian, CEFAS, pers. comm.).

Guay & Himmelman (2004) summarised the beneficial effects of shell on the seabed, stating that species richness, diversity and abundance in marine communities generally increases with increasing substratum heterogeneity. This is generally favourable to conservation value. Larval settlement is often favoured as a result of the increased availability of surfaces to settle upon, increased larval retention due to increased bed roughness and reduced current velocity and increased availability of suspended food particles (again, related to the lower current velocity). A heterogeneous bed also provides refuge from predators and high current velocities and may reduce interspecific competition. Furthermore, Guay & Himmelman (2004) suggested that the removal of scallops and potential shell litter from scallop beds could have a negative impact on scallop populations. Shell material often supports the growth of red algae and filamentous hydrozoans which provide favourable substrata for larval settlement (e.g., bivalve spat); empty shells provide protection for scallops against predation by crabs and the reduction in current velocity caused by the increased bed roughness and/or protection offered by the shells themselves allows increased feeding periods for filter feeding organisms, thus increasing growth rate. It is likely that other species may also benefit from the habitat provided by shell litter (Gutiérrez *et al.*, 2003).

Guay & Himmelman (2004) also examined the effect of adding shell litter (from *Chlamys islandica*) to both sandy and rocky beds and found species diversity to increase in both habitats. This effect was greatest in sandy areas, where the number of species increased by a factor of 3.7 and diversity increased by a factor of 1.9; this was attributed to immigration rather than larval settlement. The effect of adding shell varied among the different invertebrate species with the abundance of scallops (*Chlamys islandica*) increasing in rocky habitats (where they are naturally present) to a greater extent than in sandy habitats (where they are mostly absent).

Overall, it was concluded by Guay & Himmelman (2004) that the addition of shell to the seabed would have a beneficial effect on the epibenthic community, including commercial species such as scallops, whelks and urchins. However, problems may arise if shell material is disposed of inappropriately. That is, disposing of large amounts of material in a low energy area may cause smothering of the sea bed and the infaunal, sedentary organisms inhabiting it. Guay & Himmelman (2004) stated that the impact of the addition of shell to the seabed on infaunal species was unknown. However, there is evidence that, in muddy habitats, the addition of shell can cause mortality of certain species. For example, Iribarne *et al*. (1995) demonstrated increased mortality of the infaunal bivalve *Macoma balthica* following the addition of oyster shell to muddy habitats in the Grays Harbour estuary, Washington State. This was attributed, in part, to increased predation pressure by Dungeness crabs (*Cancer magister*) which readily colonised the shell habitat. It was concluded that artificial shell habitat can enhance the settlement and survival of some species but may adversely affect the ecology of non-target species through changes in predator-prey dynamics. Similarly, the smothering of muddy sediments with oyster shell was also found to reduce recruitment of the ghost shrimp *Neotrypaea californiensis* (Feldman *et al*., 1997).

Where scallops are shucked at port (an activity which is not permitted in the U.K., the muscle and remaining shell are often thrown overboard leading to problems with shell accumulation along wharves (Guay & Himmelman, 2004). As harbours are naturally low energy areas, there may be concomitant water quality problems. Shell waste has been disposed of for a number of years over the cliff at New Quay in Cardigan Bay, Wales and this has led to concerns about its impacts on, for example, dolphin populations. Furthermore, accumulation of this material has affected the amenity value of the beach (Dr. C.M.G. Vivian, CEFAS; J. Higgins, CCW, pers. comm.).

Changes to the sediment characteristics of the receiving environment may occur if the local sediments do not contain significant amounts of shell. Even if the hydrodynamic conditions allow dispersal, some shell is likely to be incorporated into the sediment which may impact upon the benthic infauna leading to overall community change. In turn, this will impact upon predatory species. If the aim is to use waste shell in a beneficial manner, such as habitat creation or restoration, then it must be disposed of in an appropriate environment. Habitat creation would not be achieved by dumping shell in a high energy area where it would be widely dispersed. Consideration should also be given to the target species, i.e. those which are to benefit from the habitat, with sandy sites being best for whelks and rocky sites being best for enhancing scallop populations (Guay & Himmelman, 2004). Conversely, if the aim is simply disposal, such as in the absence of an area where the shell could be of benefit, then a highly dispersive site should be chosen. Clearly low energy, muddy habitats should be avoided due to the adverse effects on the benthic community reported by Feldman *et al*. (1997) and Iribarne *et al*. (1995). In all cases, characterisation of the benthic community in the proposed disposal area is required in order to identify the presence of any infaunal or epifaunal species which may be adversely affected by the addition of shell material.

### 4.2.1. BENEFICIAL USES OF SHELL MATERIAL

The disposal of shell in the marine environment may also play a role in coastal defence and erosion reduction in intertidal areas. Meyer *et al*. (1997) added oyster (*Crassostrea viginica*) shell, or 'cultch', to the fringes of *Spartina alterniflora* marshes (smooth cord grass, a habitat in which oyster reef naturally occurs) in North Carolina, USA and monitored erosion and marsh edge vegetation stability. The presence of the cultch was found to substantially reduce wave action and erosion along the edge of the marshes, resulting in increased sediment stabilisation and a reduction in the loss of marsh edge vegetation. In addition to this wave dampening effect, cultch also provides an important faunal habitat with numerous economically important species being associated with oyster reefs (Wells, 1961). Meyer *et al*. (1997) and Coen *et al*. (2000) stated that such habitat was under threat as a result of pollution and overfishing and that the deliberate creation of reefs using shell would create valuable habitat for these species.

The beneficial effects of shell are dependent upon various factors. Gutiérrez *et al*. (2003) found that individual shell specific properties (e.g. size, volume, texture, degree of damage) and the spatial arrangement (aggregation or dispersed) which determine the accessibility of the resource to organisms which may potentially use it. Larger shells provide a larger surface area for colonisation and therefore usually support more individuals, greater species richness and, potentially, larger organisms than do small shell fragments (Creed, 2000, in Gutiérrez *et al*., 2003). Aggregations of shells provide greater habitat heterogeneity and also increase the potential for protection against predators since the interstices between the shells are available for colonisation as well as the shell cavities themselves (Gutiérrez *et al*., 2003).

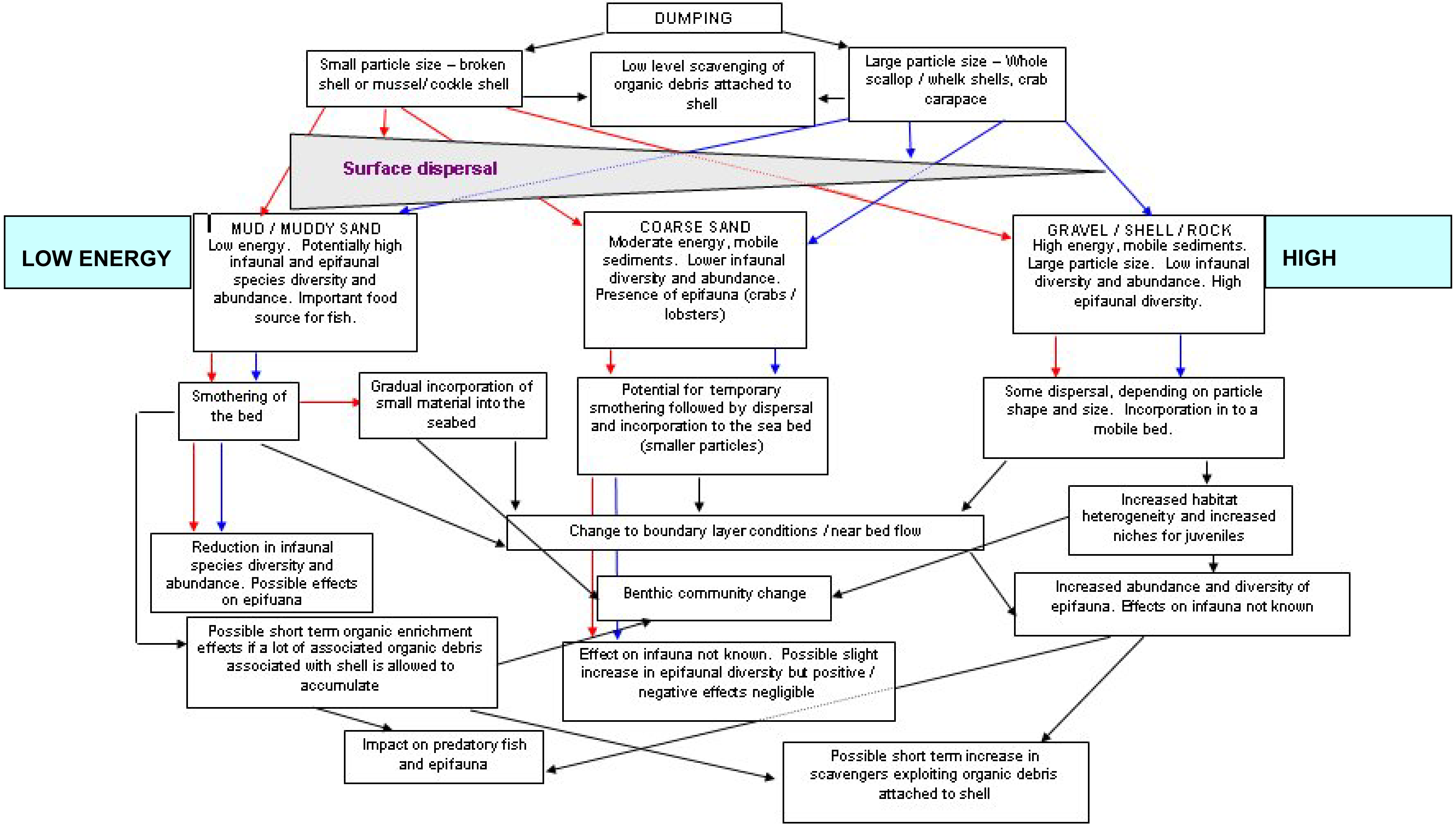


**Figure 7. Effects of dumping shell material. Thickness of grey arrow indicates the magnitude of a potential effect. Blue arrows indicate fate of large particles, red arrows small particles and black arrows both.**

## 4.3. Effects on scavenger populations

Certain types of waste could remain at or near the surface for up to six hours where it may be available to scavenging birds (Blaber 1995; Hüpppop & Wurm 2000). The removal and dispersion of this floating material is dependent upon the number of surface scavengers together with wind and wave-generated water circulation patterns. Surface phenomena such as Langmuir circulation ensures maximum dispersion when the wind is strong in relation to the current and blowing perpendicular to the direction of the current (Smith & Thorpe, 1999).

Whilst sea birds play an important role in the removal of this floating waste, the presence of the waste plays an important role in sustaining the bird populations (Furness *et al*., 1992). In the North Sea, species which particularly benefit from discards from fishing boats include fulmar (*Fulmarus glacialis*), gannet (*Sula bassana*), great skua (*Stercorarius skua*), great black-backed gull (*Larus marinus*), lesser black-backed gull (*L. fuscus*) and herring gull (*L. argentatus*) (Furness *et al*., 1992). Camphuysen *et al* (1995, in Bluhm & Bechtel, 2003) reported up to 66% of the North Sea avifauna to be feeding on waste seafood during the summer, consuming around 255,000 tonnes of discard (unprocessed waste from trawlers) and 55,000 tonnes of offal (processed seafood waste). This was equivalent to around 39% of the total available fishery waste and it was estimated that this waste could potentially support over $6 \times 10^6$ birds. Hüpppop & Wurm (2000) reported that over 70% of the pellets from herring gulls and great black backed gulls feeding around Helgoland (German Bight, North Sea) were composed exclusively of fishery discard remains during trawling periods. During periods when trawlers were not operating, the populations of these two species declined by over 80% and with their body mass decreasing by up to 25% (Garthe *et al*., 1996). Wahl & Heinemann (1979, in Bluhm & Bechtel, 2003) reported a notable increase in the number of birds around fishing vessels in comparison to unfished areas.

The size, weight and type of the waste taken by scavenging sea birds is species specific although a wide range of waste items are taken by birds. Furness *et al*. (1992) reported the size of discarded fish/shellfish consumed by seabirds to range from 5-35 cm. However, Garthe & Hüpppop (1994) found over 80% of roundfish (particularly Gadidae and Clupeidae) to be taken by herring gulls, terns and black headed gulls in the North Sea whilst only 8% of flatfish were taken, presumably because of their shape. Bertellotti & Yorio (2000) noted that those discards rejected by scavenging sea birds were generally deep bodied with respect to their length, flatfish and those with strong dorsal fin spines or caudal spines, making them difficult to handle and swallow. These studies suggest that there is much of the dumped waste is consumed before it reaches the sea bed. However, given the impact of the increased food availability on bird populations, coastal disposal sites close to inhabited or tourist areas should not be used because of nuisance created by large numbers of sea birds. Furthermore, large populations of birds could lead to increased concentrations of faecal coliforms in the feeding area (AMEC, 2003).

Whilst changes to the habitat characteristics impact upon the bacterial, macrofaunal, megafaunal and fish populations, Champ *et al.* (1981), Bluhm & Bechtel (2003) and AMEC (2003) suggested that the disposal of seafood processing waste at sea may also lead to changes in the abundance and composition of scavenging species on the sea bed. It is well documented that mobile scavengers and predators are attracted to areas impacted by demersal trawling gear (Kaiser & Spencer, 1994, Jennings & Kaiser, 1998) and it has been suggested that the discards from trawlers are beneficial to populations of scavengers (Ramsay *et al.,* 1996; Veale *et al.,* 2000). Species attracted to discards included *Asterias rubens, Astropecten irregularis,* (starfish) *Liocarcinus* spp., *Pagurus* spp. (crabs) (Veale *et al*., 2000), *Cancer pagurus* (edible crab), brittlestars, flatfish and gadoids (Jenkins *et al.*, 2004), *Crangon crangon* (brown shrimp)*, Buccinium undatum* (common whelk)*, Carcinus maenas* (shore crab)*, Limanda limanda* (dab) and *Merlangius merlangus* (whiting) (Ramsay *et al*., 1997). Groenewold (2000, in Bluhm & Bechtel, 2003) found 46 benthic species to be attracted to baited traps in the North Sea although 70-80% of the organisms were hermit crabs, swimming crabs and starfish. Other common groups included amphipods (particularly lysianassoids), shrimps, whelks, and fish (predominantly gadoids). However, Jenkins *et al*. (2004) stated that the degree of attraction was influenced by tidal flow, bait type and species interactions. Ramsay *et al.* (1998) found attraction to discards to be species specific and variable between habitats. Veale *et al.* (2000) found a correlation between water current and the arrival of scavengers (principally *A. rubens*) and suggested that water current played an important role in the distribution of carrion odours.

Jenkins *et al.* (2004) examined the response of scavengers to areas baited with *Pecten maximus* (King or great scallop) with varying degrees of damage (none, low, high). Nineteen taxa were observed during the baiting periods in comparison to the background level of nine taxa. The dominant species attracted to the site included *C. pagurus, Ophiocomina nigra* (brittlestar), flatfish (including *Pleuronectes platessa* (plaice)) and gadoids, all of which appeared at sites baited with highly damaged scallops within less than 5 hours of baiting. Kaiser & Spencer (1996) recorded a peak in the number of scavengers within 8-14 hours of baiting. At sites baited with highly damaged bait, all scallops were eaten over a period of 24 hours and it was concluded that badly damaged scallops provided a readily available food source for all scavenging/predatory species. Seafood processing waste consists of large amounts of readily available food and the results of Jenkins *et al*. (2004) therefore suggest that removal of waste from disposal sites will be rapid. It is acknowledged that species such as *P. maximus* are unlikely to make up a significant proportion of the waste generated by the seafood processing industry. However, Veale *et al*. (2000) found scavenger species to be attracted to a variety of bait.

Demestre *et al.* (2000) studied the behaviour of scavengers and predatory species in response to otter trawling in the Mediterranean Sea. Scavenging species were attracted to the trawl sites, presumably by the odour of carrion or those displaced by the trawl, increasing in abundance over time during trawling. However, this aggregative response was short-lived (several days), indicating rapid consumption of the increased food availability caused by the trawling. This, again, suggests that scavenging species may rapidly remove organic waste from disposal sites.

In contrast, Ramsay *et al*. (1998) found that scavengers were not necessarily always attracted to discards and attributed this to possible food availability prior to trawling. It was suggested that in areas where food was limited, the presence of discards may encourage scavengers to move in and exploit the plentiful food supply. Conversely, areas where food availability was always high, scavengers would have little need to migrate to exploit the discards. This may well be of relevance to the fate of seafood processing waste following disposal.

The activity of scavengers in organically enriched areas has been found to increase the rate of removal and oxidation of organic material as a result of consumption, bioturbation and, hence increased oxygen supply to buried organic matter, and resuspension of detritus (Katz *et al.*, 2002). Katz *et al*. (2002) and Lupatsch *et al*. (2003) found the bottom feeding grey mullet *Mugil cephalus* efficiently remove a significant amount of organic matter from the sea bed beneath fish cages and it was suggested that the deployment of detritivores below fish farms may alleviate the impacts of aquaculture. Whilst this study did not provide evidence that such species were attracted to organically enriched areas, the findings may be of relevance to the disposal of seafood processing waste.

# 5. CASE STUDIES

## 5.1. Kodiak Bay, Alaska (Stevens & Haaga, 1992)

Seafood processing generates a large amount of waste in Alaska with approximately 400 tonnes of waste (heads, viscera, bones) per day being generated in Kodiak Bay. The local fishmeal plant had the capacity to deal with around half of this waste such that 183 tonnes/day were dumped at sea, resulting in an annual figure of 67,000 tonnes. The USEPA permitted fish to be ground up and dumped by barge in a designated area of Chiniak Bay in waters of >91 m. Dumping was carried out by a combination of pumping and opening of rear gates, from a moving barge, over an area of 27 $km^2$. Whilst organic debris piles on the seabed and oxygen depletion in the water column had been reported at sites seafood disposal sites in Unalaska Bay, the EPA recognised that closure of these dumping grounds would have had an immediate and direct effect on the local economy.

Stevens & Haaga (1992) monitored the fate of this waste during dumping operations, using a two person submersible DSRV (Deep Submergence Rescue Vehicle), carried out between April and September, 1990. Monitoring was carried out at sites in Chiniak Bay and at control sites in Monashka Bay and involved visual observations (including video footage) of the dumping process, the abundance and accumulation of waste, the visible epibenthic fauna and the sediment characteristics. Grab sampling was also carried out to give quantitative data regarding the sediments.

Overall, the control site and the dump site at Chiniak Bay had very similar sediment characteristics, with a mix of shell, gravel and sand. In general, samples from Chiniak Bay contained more shell, suggesting that it was not being dispersed, and fish bone (up to 14% at some sites) with one site also containing a large amount of silt (>75%). Despite this, there were no differences in pH or dissolved oxygen concentration of the near bottom water between sites. Furthermore, there were no differences between sites within Chiniak Bay irrespective of whether waste was present.

Observations during dumping showed that large, heavy material, such as whole fish or heads, sank to the bed rapidly with viscera, skin, gills and fins (5-25 cm) settling over a period of approximately ½ an hour and smaller material taking over 1 hour to settle. Debris was seen to settle in patches (approximately 10 x 30 m), where it covered the bed completely, although these patches were separated by areas of less dense debris. This effect was considered to be site specific and dependent upon the local hydrodynamic conditions. Bacterial or fungal mats were seen around the decomposing debris, as were bones, completely stripped of flesh, although the accumulation of organic debris was not observed in any location except during the short term following dumping. Buoyant material attracted, and was scavenged, by large numbers of kittiwake and seagulls at the surface. Scavenging was not observed in the water column but, on the bed, flounder, sculpins and amphipods appeared immediately to start consuming the waste; octopus and sea stars were seen to consume debris which was several days old. In general, decomposition or consumption by scavengers was rapid with the quantity of waste on the sea bed decreasing exponentially with a half life of 8.7 days and no waste remaining 33 days after dumping.

In terms of the epifaunal community composition, the number of species and abundance were similar between the control and the dump sites and were considered typical of a stable, mature community. Whilst there were slight differences in species composition between the two bays, there was no indication that these differences were biased towards enrichment or impoverishment.

Whilst no evidence of adverse environmental impacts were detected in this study, the authors acknowledged that the data were not definitive and may have been circumstantial and should therefore be treated cautiously. That is, no sampling was carried out prior to dumping so although the study allowed spatial comparison between the two bays, no temporal comparison could be made. Therefore it is not known whether or not the disposal of seafood processing waste had caused significant change to the sea bed and the associated epibenthic communities in Chiniak Bay. Despite this, it should be noted that poor water quality and community impoverishment were not recorded and that, even with sufficient spatial and temporal sampling, the effects of anthropogenic change are difficult to distinguish from natural fluctuations. Furthermore, the USEPA considered deep sea dumping to be preferable to discharge via a pipeline which had been found to be associated with waste accumulation, poor water quality and degradation of the benthic habitat.

## 5.2. Bering Sea (Bluhm & Bechtel, 2003).

Bluhm & Bechtel (2003) studied the impacts of fish processing waste on the community structure at the surface, throughout the water column and on the seabed. This study provided indicative evidence from the North Sea, the Mediterranean Sea and Australian waters to demonstrate the fate of discarding fish processing waste at sea. This evidence was used to develop scenarios regarding the potential fate and effects on the marine environment of discarding offshore fisheries waste in the South-eastern Bering Sea.

It is stated that around $0.4 \times 10^6$ tonnes of processing wastes are produced each year in Alaska with around 95% of that originating from the Bering Sea. 85% of the offal, defined as waste produced from processing as opposed to discards from trawlers, is currently discharged at sea. It is suggested that increases in benthic biomass and community structure were observed in the Bering Sea in the early 1980s, although Bluhm & Bechtel (2003) concluded that the effect of dumping this volume of processing waste on the Bering sea is yet to be determined. In general, it was considered that the impacts of discarding waste are thought to be less in offshore rather than nearshore waters, as long as the waste is dispersed over a wide geographic area and is not allowed to accumulate and become concentrated in any particular location.

## 5.3. Peter Pan Seafoods (USEPA, 1998).

In King Cove, Alaska, commercial fishing and seafood processing contribute significantly to the local economy. The seafood processing company, Peter Pan Seafoods, processes locally harvested fish and shellfish and subsequently disposes of the waste in King Cove. However, in 1998 the Alaska Department of Environmental Conservation (ADEC) stated that disposal of this waste had led to the formation of a waste pile of solid residue measuring 0.04 $km^2$ in area with an average depth of 1 metre (ADEC, 1998; Enviro-Tech Diving, 1998). This was considered to exceed water quality standards for residues and was found to have an adverse impact on the growth of fish and shellfish and on local aquatic life. The EPA therefore carried out an assessment of the amount of settleable solid seafood processing waste which, following discharge, would not lead to significant deposition over a 0.4ha and a

0.8ha site. In this case, significant deposition was considered to be detection at greater than 300 $m^2$ and deposition of more than 1.1 cm.

In order to determine the area of deposition resulting from such a discharge, a number of biological, physical and chemical factors must be considered in order to determine the fate of the waste. These include microbial decay which is dependent on oxygen, temperature, organic matter and microbial biomass, the presence of scavengers, the chemical composition of the waste such as protein, fats, carbohydrates and soluble organic compounds and the amount of skeletal/shell material and connective tissue. Decomposition and dispersal is also controlled by water column stratification, the strength of tidal and wind induced currents, water temperature and sea floor topography.

Macerated solid waste discharged (via a pipe) by Peter Pan Seafoods was screened prior to discharge to remove solids of >1mm. The waste was discharged to an area of 15 m depth with currents of 5 cm $s^{-1}$ or less. The EPA modelled the environmental characteristics and calculated that 227 tonnes $yr^{-1}$ (or 0.62 tonnes per day) could be discharged to this area without causing a zone of deposition (ZOD). However, applying a 10% margin of safety to these values gave a total annual figure of 200 tonnes and a daily figure of 0.56 tonnes as a total maximum daily load (TMDL). These figures were derived assuming a particle size of <1mm.

Estimates of the settling velocity and resuspension current speeds were also given for various seafood waste particle diameters (and particle densities). For example, it was estimated that a particle of seafood with a 1mm diameter and a density of 1.13 g $cm^{-3}$, would settle at a rate of 0.017 m $s^{-1}$ whereas a particle of similar density but of 10 mm diameter would settle at a rate of 0.165 $ms^{-1}$, almost an order of magnitude higher. When comparing the current speed required to resuspend these two particles, it was estimated that the 1mm diameter particle would be resuspended with a resuspension current speed of 0.11 m $s^{-1}$ whereas a resuspension current speed of 0.4 m $s^{-1}$ (nearly 4 times stronger) would be required to resuspend the larger particle (10 mm in diameter). Although these estimations give an indication of settling velocities and resuspension current speeds, it is considered that these figures are likely to be site specific.

## 5.4. Ketchikan, Alaska (Tetra Tech, 2004).

During an eight month (August 2003 - April 2004) study, the Alaska Department of Environmental Conservation (ADEC) monitored the dispersion and decomposition of seafood processing waste piles situated in the Tongass narrows ,approximately 30 m off the Ketchikan waterfront, Alaska (Tetra Tech, 2004). Both video and diver surveys were carried out to determine the relative changes in area and volume, with time, of seafood processing waste piles, consisting of different particle sizes of waste, together with the presence of fish and invertebrate species on or near the piles. Material was dumped at water depths ranging from 4 to 6 m with 4.5 tonnes of 1.3 cm diameter and 2.5 – 5 cm diameter waste being deposited and a further pile consisting of fish heads (1.4 $m^3$). The substratum in the area was clean with minimal debris.

During the eight-month study, all waste piles decreased in volume as a result of compression and collapse due to settlement and decomposition, dispersal by tidal currents and foraging by scavengers. The greatest reduction in volume and area were noted from waste piles composed of 2.5 – 5 cm and 1.3 cm material with the greatest degree of dispersal (i.e. increase in pile area) being associated with the smallest particle sizes. It was recommended that 2.5 – 5 cm sized particles had the potential for rapid decomposition with limited spreading which would be ideal for an area where the spreading of waste was a concern. However, decomposition rates and the potential for dispersal were much greater for the 1.3 cm particles suggesting that this particle size may be more appropriate if a dispersive, rather than aggregative, disposal method were to be used.

It should be noted that, in this study, waste was dumped deliberately to form piles. Several months later, these were still evident, although greatly reduced in volume and it is therefore suggested that dumping should take place in small amounts over a widespread area.

No conclusions could be drawn regarding the effect of particle size on the organisms attracted to the waste. In general, macroinvertebrates were seen at the edges of the waste piles with fish being more commonly seen in the water column above or at the periphery of the waste piles. No organisms were seen on the waste piles themselves although bacterial and fungal mats were observed.

## 5.5. Cordova Fisheries Enhancement Project, 2004-2006 (K. George, Alaska Department of Environmental Conservation, pers. comm.).

Cordova fish processors are currently limited to the amount of filleted fish they can produce as a result of the amount of waste associated with it. That is, 80% is wasted following filleting in comparison to the 20% (heads and guts) associated with the production of whole frozen fish. During 2004, this limit on waste production and discharge prevented processors from filleting and although a fishmeal plant will become operational during 2005, only 50% of the seafood waste will be processed.

The Prince William Sound Science Centre (PWSSC) together with the Alaska Department of Fish and Game (ADF&G), the Alaska Department of Environmental Conservation (ADEC) and Cordova Seafood, have proposed a three year study (Cordova Fisheries Enhancement Project, 2004-2006) to determine the rate of decay and consumption of seafood waste dumped in the Northern Orca Inlet, Alaska (K. George, ADEC, pers. comm.). It is proposed that, if placed in the right area, the waste could provide a beneficial food source for aquatic life and will therefore be rapidly removed from the sea bed through the processes of consumption and decay.

In 1975, the United State Environment Protection Agency (USEPA) stated that, prior to discharge (via a pipeline), waste from seafood processing plants must be ground to 1.3 cm in size. This was to ensure availability to a wide range of scavengers and to destroy the swim bladder in order to prevent floating. This resulted in a decrease in odour and in fish carcasses being washed up on beaches. However, there were a number of adverse effects including an increase in the number of gulls in the area, causing nuisance and contamination of drinking water sources (by bacteria associated with droppings), waste accumulation which reduced the rate of decomposition, adverse impacts on clams, fish and crabs, parasitic infection in sea otters leading to death and benthic community effects.

This proposal outlines methods to compare the ecosystem response in and around sites with either 1.3 cm size waste particles or heads and guts and is designed to expand upon the small scale study carried out in Ketchikan (Tetra Tech, 2004). 18 – 23 tonnes of salmon waste is to be distributed, from a slowly moving barge, twice a week for a period of six weeks in the Northern Orca Inlet (80 m depth). Current velocity in this area is low and settlement of waste on the seabed is expected to occur over an area 0.2 $km^2$ with dispersal over 0.7 $km^2$. Following sampling of the fish, benthic invertebrates and birds, a conceptual model is to be developed to demonstrate the fate of ground and unground waste and to determine whether or not the effects of disposal might be beneficial. This project is still at the proposal stage but, should it go ahead, the findings will be of relevance to future waste management schemes elsewhere.

# 6. IMPACT ASSESSMENT AND MANAGEMENT

Following the above quantification of the potential problem, it is necessary to discuss the scientific approach to detecting impacts (Section 6.1) and the means by which this approach is applied to site selection and impact assessment (Section 6.2). In essence the process involves setting up Environmental Quality Objectives, which may relate to amenity and socio-economic aspects as well as scientific ones and designing the disposal and monitoring regime to meet these requirements. Site selection is a key element in this process.

## 6.1. Scientific approach to detecting environmental impacts.

Prior to the disposal of waste at sea (or any other activity which has the potential to cause adverse impacts), an assessment of the consequences on the receiving environment should be made and summarised as a series of ‘impact hypotheses’.

Following the disposal of any type of waste at sea, it is necessary to demonstrate that the environment has not been adversely affected as a result. That is, there must be no unacceptable loss of quality or deterioration to the health of the system in its structure or functioning or hindrance to the uses and users of the area (MEMG, 2003). The requirement for monitoring following the initial impact assessment is determined on a case by case basis although past experience of the disposal of fish processing waste at sea has suggested that monitoring is unnecessary in most cases (OSPAR, 1998). However, this must be confirmed by an initial impact assessment. The present section outlines the scientific approach to the detection of impacts on the marine environment

Monitoring of the marine environment may involve physical, chemical and biological measurements and is generally carried out in order to confirm compliance with licence conditions and/or determine impacts (MEMG, 2003). As stated by MEMG (2003), the actual, potential or perceived effects of waste disposal will dictate the nature and magnitude of the monitoring required.

It is of value to consider the potential changes against the Environmental Quality Objectives (Table 3) adopted by the Group Coordinating Sea Disposal Monitoring of the Marine Pollution Monitoring Management group (now Group Coordinating Seabed Disturbance Monitoring of Marine Environmental Management Group). Whereas most of the above impacts of disposal relate to the ecological system, the resultant impacts on the uses and users of the marine environment are often of greater prominence and more public concern. These include the actual or perceived effects on socio-economic aspects such as fisheries, and aesthetic aspects including recreation and tourism. Similarly, the perceived or actual effects on the conservation importance of an area will be of concern, especially where the habitats and species within and adjacent to the disposal areas are of importance. It is considered here that the perception of an effect has to be regarded as important as the detection of an actual effect and thus monitoring will be required to allay such concerns.

**Table 3. Environmental Quality Objectives as derived by GCSDM.**

| Use | Objective |
|---|---|
| Amenity use | Maintenance of environmental quality, so as to prevent public nuisance arising from aesthetic problems and interference with other legitimate users of the sea |
| Commercial harvesting of fish and shellfish for public consumption | Maintenance of environmental quality, such that commercial marine fish and shellfish quality shall be acceptable for human consumption, as determined by fisheries legislation such as the Shellfish Hygiene Directive and Shellfish Waters Directive |
| Protection of commercial species | Preservation of the general well-being of commercially-exploited species |
| General ecosystem conservation | Maintenance of environmental quality to prevent deterioration of aquatic life and dependent non-aquatic organisms within the existing ecosystem of the area |
| Preservation of the natural environment | Impacts shall be restricted to the designated disposal zone, areas outside of which shall be non-impacted reflecting the quality of the adjacent estuarine or marine environment |

#### POTENTIAL IMPACTS

The potential impacts of the disposal of seafood processing waste at sea are discussed above in Sections 4 and 5 based on a small number of case studies together with various studies carried out in relation to the discharge of liquid waste from the seafood processing industry, fish farming and the disposal of sewage sludge and dredged material. The impacts of the disposal of fish and shellfish waste can affect the water column features, the physical and chemical nature of the seabed, and the seabed biota. The potential impacts could be manifested in the following:

- a deterioration in the overall health/quality of the marine ecosystem;
- a reduction in the socio-economic aspects of the sea including fishery and amenity interests;
- an interference with the legitimate uses (i.e. those legally permitted) of the sea/recreation and navigational aspects;
- a reduction in the aesthetic qualities associated with the area. These main features can be interpreted against a set of Environmental Quality Objectives derived for the marine environment (MEMG, 2003).

The impacts associated with these activities require both near-field and far-field consideration (MEMG, 2003). That is, there may be impacts in the immediate area receiving the waste (near-field) but also away from the dumping ground (far-field) into which deposited material might be carried and re-deposited. Similarly, there is a temporal aspect to the impact in that waste may accumulate and cause long term impacts or disperse rapidly with only short term impacts (McLusky & Elliott, 2004).

#### SAMPLING STRATEGY

Any sampling strategy must be designed to ensure that environmental change can be attributed to a cause (in this case, waste disposal) rather than to natural or unexplained variability (Underwood, 1994; MEMG, 2003). Determination of the impacts of pollution on the environment must be based on comparative analysis between the status of a reference (non-polluted) site and the site influenced by the waste materials (Prevost, 1999). The BACI (Before After Control Impact) approach, or a variation of it, based on the sampling strategy in terms of spatial and temporal monitoring, is now widely accepted as an appropriate means of detecting human impacts on the environment (Underwood, 1994; Ellis & Schneider 1997).

Initially, the determination of environmental impact using the BACI approach involved monitoring at an impacted site and at a control site before and after the influence of a factor which may cause environmental change (Underwood, 1994). However, without replicated control sites, differences between locations cannot unambiguously be attributed to a disturbance due to pollution. Similarly, any change along a gradient from a point source of pollution needs to be contrasted with a similar gradient at a non-impacted site (Ellis & Schneider, 1997; Lardicci *et al.*, 1999). Changes along an impacted gradient should also be compared with changes along a spatial gradient which is not influenced by pollution (Lardicci *et al.*, 1999). Therefore, Beyond BACI designs which use one impacted site together with several control sites are now recommended for the reliable detection of environmental impacts. Such designs allow differentiation between natural background variability and that caused by pollution or some other environmental impact (Chapman *et al.*, 1995). This is particularly important as variability within communities at impacted sites is much greater than that at control sites (Warwick & Clarke, 1993).

Most approaches to determining the impact of a pollutant involve the analysis of community composition and the determination of any change from that expected (McManus & Pauly, 1990; Elliott, 1994; Little, 2000; McLusky & Elliott, 2004). Benthic invertebrate communities are commonly used as a means of detecting ecological impacts on the environment since they are highly susceptible to pollution or other forms of environmental change (e.g. changes to the substratum, changes to the hydrodynamic regime). Many of them are sessile or migrate only small distances during their life time (Bamber & Spencer, 1994 in Lardicci *et al.*, 1999). Changes to benthic communities can therefore directly reflect changes in environmental conditions. Warwick (1993) stated that where organisms cannot be identified to species level, data analysis can be carried out at higher taxonomic levels without losing information, although this approach has not been used for the detection of subtle changes. Furthermore, there are now broad but reasonably well-defined patterns, in terms of the effects on numbers of species, abundance and biomass, which invertebrate communities follow in response to certain types (mainly organic) of pollution (Pearson & Rosenberg, 1978). The impacts of the disposal of seafood processing waste are likely to include changes to:

- the particle size distribution, organic content and redox condition of the sediments in the receiving environment;
- the benthic and epibenthic (e.g. scavengers) community structure;
- the fish populations;
- the water quality in terms of dissolved oxygen, pH, turbidity/suspended solids, particulate and dissolved organic carbon concentration and nutrient concentrations.

As a minimum, standard sampling techniques would include depth profile measurements for dissolved oxygen, pH and turbidity. Macrofaunal and sediment sampling would be carried out using a grab (the type depending on the substratum type) and would be followed by identification, enumeration and biomass determination of the organisms in the samples. Underwater video survey techniques would provide a visual indication of the species feeding at the site and the overall environmental condition together with the rate and extent of degradation and dispersion of the waste.

Characterisation of the sediments would include particle size analysis together with determination of the organic content (expressed as % loss on ignition). Finally, beam trawling could be carried out in order to detect impacts on the epibenthic and demersal fish populations. Full details of marine environmental sampling methods are given in Holme & McIntyre (1984), MEMG (2003) and McLusky & Elliott (2004).

A robust sampling strategy would include spatial and temporal monitoring with replication to allow statistical analysis. However, economic limitations must be considered and spatial coverage should not be compromised by replication at a small number of sites. A more advisable approach would involve key stations where replicate samples were taken (for statistical robustness) with a number of stations where only one sample was taken, thus allowing greater spatial coverage without incurring excessive cost.

**DATA INTERPRETATION**

The detection of benthic community change is carried out using various univariate and multivariate statistical techniques outlined in Elliott (1994). Simple, commonly used univariate techniques for detecting disturbance include recording the number of species (S), abundance/unit area (A), biomass/unit area (B), A/S and B/A ratios where a high A/S value together with a low B/A value would be indicative of stress (Gray *et al.*, 1988). That is, the majority of the individuals would belong to very few species and the community would be composed of organisms of small body size. Diversity indices are also commonly used as a means of comparing communities and detecting stress (Krebs, 1980) and usually include the Shannon-Weiner index of diversity (H') and Pielou index of evenness (J') (Pielou, 1975; Clarke & Warwick, 1994). Such primary and derived statistics are calculated for each sample with mean values being compared on a spatial and temporal basis by means of analysis of variance (ANOVA).

Univariate statistics are useful in that they give an absolute or derived value which can be used to determine whether or not community differences can be attributed to conditions causing stress (Gray, 1988). This can then be used to make an assessment of the severity of the change on a spatial and temporal scale (Warwick & Clarke, 1991). However, these techniques are not species specific and two communities with entirely different taxonomic compositions could appear to have the same structure using these techniques alone (Warwick & Clarke, 1991). Therefore, these techniques are used in combination with multivariate analysis which compare communities on the basis of their component species and their relative importance in terms of abundance and biomass. Such methods have repeatedly been found to be more sensitive in discriminating between sites and times than univariate methods (Warwick & Clarke, 1991). A commonly used technique is the calculation of the Bray-Curtis similarity coefficient (also known as the Czekanowski coefficient). Similarity matrices can then be represented graphically in the form of dendograms which allow the identification of distinct groups of samples or sites with distinct community structures, implying that the different patterns of the species present and their abundance or biomass occur consistently within the different groups (Clarke & Warwick, 1994).

Whilst being more sensitive than univariate techniques, most multivariate techniques generate rather than test hypotheses and may give no indication of the underlying cause of the species differences between sites. It is therefore recommended that they are used to complement rather than replace univariate methods (Gray *et al.*, 1988; Elliott, 1994).

Graphical techniques include the use of the log-normal distribution (Gray, 1979; 1981) and the use of *k*-dominance curves (Lambshead *et al.,* 1983), where cumulative percent dominance (in terms of abundance or biomass) is plotted against the species rank, on a logarithmic scale (Warwick, 1986). Warwick (1986) proposed a variation on the use of these curves whereby the abundance and biomass curves were overlaid on the same graph. This method is known as the ABC (abundance-biomass comparison) method and is based on the assumption that unstressed or stable environments are characterised by one, or few large species, each represented by few individuals. In contrast, stressed communities are characterised by high numbers of short lived *r*-strategists with small body size, a high reproductive capacity and a variable population size. The difference between the abundance and biomass k-dominance curve is represented by the W-statistic. The ABC method, together with the SAB method (Pearson & Rosenberg, 1978), indicates the value of assessing the biomass of whole communities or for individual taxa (Elliott, 1993; Elliott, 1994).

## 6.2. Site selection

While there are other discussion documents which outline the considerations inherent in site selection for dredged material disposal (Vivian 1992, MEMG 2003), the OSPAR (1998) document is the only previous policy document for fish and shellfish waste disposal. These documents therefore indicate general philosophies which require to be adopted for the disposal of fish and shellfish waste. Those general philosophies are therefore used by the licensing authorities prior to permitting disposal. In summary, in site selection, there is the need:

- to minimise interference with uses and users,
- to minimise the environmental impact,
- to evaluate options for disposal,
- to determine the capacity of the disposal site,
- to characterise the receiving environment,
- to determine the transport of material thus influencing near and far field effects,
- to determine the accumulating/dispersing nature of the site,
- and to consider the acceptability of any effects.

These aspects are considered in further detail below.

### 6.2.1. PRE-DISPOSAL CONSIDERATIONS.

Whilst the disposal of seafood processing waste at sea is exempt from the general prohibition of waste disposal at sea, there is potential for adverse effects on subtidal and intertidal environments. In cases where beneficial uses for the waste cannot be found, careful selection of the dump site is imperative to in order to avoid environmental problems (OSPAR, 1998). The fate of seafood processing waste in the sea is dependent on the nature of the waste including the size of the pieces (heads, viscera, shell fragments etc), vessel practices and mode of dumping, and the physical and biological characteristics of the dumping site, including sediment type/particle size, hydrodynamic regime, infaunal and epifaunal communities and the presence of scavengers (Bluhm & Bechtel, 2003).

#### INFORMATION REQUIREMENTS

The selection of a suitable site must involve consideration of the environmental characteristics (physical, chemical, biological and aesthetic aspects), together with economic and operational viability. It is necessary that other legitimate uses and users of the sea are not jeopardised as a result of disposal at sea (e.g. aquaculture or commercial fishing). This includes consideration of spawning, nursery and feeding grounds, together with the potential for interference with fishing gear. Similarly, the conservation features of an area must not be affected with sensitive habitats and those with rare, vulnerable or endangered species being considered unsuitable for waste disposal (OSPAR, 1998). Furthermore, the future uses and users of the sea must be considered and baseline and monitoring studies are essential in the evaluation of any new dumping activity (OSPAR, 1998).

OSPAR (1998) stated that the following information should be obtained for proposed dump sites before final selection:

- Physical (topography, particle size distribution), chemical (organic content, redox condition), and biological (benthic and epibenthic biota) characteristics of the sea bed;
- Physical, chemical and biological characteristics of the water column (currents, dissolved oxygen, suspended solids, pelagic species, primary productivity);
- The dumpsite must not impact upon:

  I. protected areas or critical habitats of scientific or biological importance;
  II. Mari culture operations;
  III. spawning, recruitment or nursery areas;
  IV. migration routes for marine organisms;
  V. areas of natural beauty, cultural or historical significance;
  VI. recreational areas;
  VII. fishing areas (commercial or recreational);
  VIII. shipping lanes;
  IX. engineering uses (e.g. power generation, cables and pipelines, mining/dredging areas, water intake).

In addition to these factors, it is also recommended that hydrographic data (e.g. from admiralty charts or existing data sets) be sought to give an indication of the residual current patterns and strengths, the potential for dispersion and an indication of the direction and area to which the waste material might be dispersed (see Stage 3 Disposal site Characterisation).

#### 6.2.2. APPROACH TO SITE SELECTION

Site selection must include an assessment and/or prediction of the potential effects of dumping in that area, by generating impact hypotheses, which must include the potential impacts on living resources, amenities and other legitimate uses of the sea. These impact hypotheses must indicate whether monitoring will be necessary and, if so, give a monitoring programme (OSPAR, 1998). Since the environmental response to a specific waste loading will differ with coastal areas, it is important to develop methods to estimate the environmental response to a specific waste loading (Nordvarg & Hakanson, 2002). This may be regarded as indicating the ability of an area to assimilate, degrade and disperse the waste, i.e. the assimilative capacity of the area. Because of this, the monitoring requirements, if any, will be site and case specific.

MEMG (2003) outlined a detailed procedure for the determination of monitoring for the disposal of dredged material and it outlined the considerations within that monitoring decision, where those considerations include the possibility of environmental impact. The protocol indicated for dredged material disposal can be adapted specifically for the disposal of seafood processing waste (Figure 8). A similar flow diagram was produced by MEMG (2003) to assist completion of a series of tables and matrices, indicating the rationale for site selection at each stage of the diagram. The key factors for consideration outlined by MEMG (2003) include those which specifically relate to the disposal of seafood processing waste given in OSPAR (1998). The present section outlines the physical, chemical and biological features of the waste and the receiving environment which must be considered before disposal takes place. Additionally, the socio-economic and conservation interests of the environment are to be determined.

The determination of actual or potential effects can be regarded as a set of decisions within the following stages:

**Stages 1-3;** characterisation of the waste and site.

**Stages 4-6;** Assess the suitability of the site and potential fate of the substances on the site.

**Stages 7 and 8;** discuss the design of the environmental impact assessment and the monitoring requirements.

**Stage 1 - Waste type:**

<u>a). Waste characterisation</u>

The first stage involves the determination of the origin of the waste, including whether it came from wild or farmed fish of indigenous or non-indigenous species, of local origin or imported from abroad, the determination of the process by which it was generated and the nature of any waste treatment (raw/cooked, untreated/irradiated/disinfected) (OSPAR, 1998; London Convention, undated). It is also necessary to define the amount of waste (in tonnes) to be disposed.

Characterisation of the waste must then be carried out in terms of the proportion of organic and inert, inorganic (shell, exoskeleton) material and the proportion of solids and liquids the waste is composed of and the proportions of whole and macerated fish. This should include details of the particle size, expected buoyancy or tendency to sink rapidly and an overall description of any chemical or biological contaminants which may be present (e.g. accumulation of contaminants in shell or tissue, presence of pathogens, parasites or alien species). Where the waste is predominantly composed of organic material, determination of the BOD (biological oxygen demand) and COD (chemical oxygen demand) may be required.

Full chemical characterisation of fish waste is not normally necessary unless contamination is suspected (for example Section 4.1.2 discusses potential contamination by metals and other pollutants; this is considered to be a very low risk in UK waters). However, waste treated with disinfectant or insecticides would be considered completely unsuitable for disposal at sea due to the addition of chemicals. Similarly, biological characterisation may be necessary if contamination by pathogenic bacteria and viruses and/or alien species/parasites is thought to be present. Biological contamination must always be taken into consideration (see Section 4.1.5) to prevent the transmission of disease and, in rare cases, transfer of pathogens to humans (OSPAR, 1998; London Convention, undated).

b) Disposal operation.

The method of disposal must be decided (dispersive or aggregative), the expected volume of each type of waste and the frequency of disposal, in terms of number of loads per day, the interval between dumping operations and whether dumping is to be carried out at a single site or seasonally at multiple sites (OSPAR, 1998; MEMG, 2003). OSPAR (1998) also stated that, due to changes in the characteristics of the waste whilst awaiting disposal (decomposition, ammonia and sulphide production), consideration should be given to the possible need for waste storage and preservation (e.g. deep freezing).

The aims of waste disposal should be considered before the method of dumping is chosen. That is, organic waste should ideally be dispersed as quickly as possible to ensure rapid degradation and reduced potential for impacts on the physical, chemical and biological properties of the disposal site. Therefore, a dispersive disposal technique (i.e. progressive release from a moving vessel) should be used. However, if the aim is habitat creation or restoration, by, for example, the dumping of shell, a more aggregative method (i.e. dumping over a small area) would be appropriate.

**Stage 2. Proximity of the proposed site to other legitimate uses/users of the marine environment.**

OSPAR (1998) stated that current or future legitimate uses of the sea must not be jeopardised as a result of waste disposal at sea. An assessment of the likely effects on the uses and users of both the water column and the seabed must be made in terms of the physical, chemical, biological and aesthetic value of the site. The level of risk should be defined as none, low, medium or high (MEMG, 2003). MEMG (2003) provided a list of activities which should not be adversely affected - these can be broadly classified into four groups.

a) Conservation interests

Areas with statutory conservation designations (such as Special Protection Area (SPA), Site of Special Scientific Interest (SSSI), Special Area of Conservation (SAC)) and sites of nature conservation importance should be avoided. Such areas may also include intertidal habitats such as seagrass beds and both subtidal and intertidal sites of conservation value must be identified prior to disposal. If disposal is likely to affect the conservation value of a European Marine Site, i.e. encompassing SAC and/or SPA, which occurs either outside or within such an area, the licensing authority must ensure that an 'Appropriate Assessment' is carried out in order to prove that the activity will not impact upon the interest features of the site (according to Regulation 33 guidelines of the Conservation (Natural Habitats &c.) Regulations, 1994). Similarly, sites of outstanding natural beauty, cultural or historical sites (e.g. marine archaeological sites) must not be adversely affected.

b) Fishing

Disposal should no impact commercial or recreational fishing or mariculture activities, or upon fish stocks nor should it interfere with fishing gear. Furthermore, there should be no deterioration in the environmental quality of spawning, recruitment and nursery areas and migration. Given that the attraction of scavengers is dependent upon food type and availability (Ramsay *et al.*, 1998), together with the concomitant biological and physical interactions, disposal sites should be positioned in order to maximise the potential for the biological removal and degradation of organic matter. However, it is possible that the catch rates of commercial traps could be reduced because of the waste providing increased food availability for crab and lobster. There is anecdotal evidence that this occurred in areas where purse seiners had released quantities of dead mackerel (W. Lart, Sea Fish Industry Authority, pers comm.). To overcome this it is important to maximise the dispersal of the waste and to liaise with potting interests in relation to potential disposal sites.

c). Other economic interests

Economic uses of the sea include aggregate extraction, navigation/shipping and port related activities, historical and current vessel based waste disposal practices, land-based discharges/outfalls, industrial intakes (e.g. cooling water) and pipelines. These, authorised, activities must not be inhibited as a result of the waste disposal at sea. This includes avoiding impacting former sewage and other disposal sites which may be being monitored post cessation of disposal to describe any recovery process.

d) Public interest and acceptability of disposal

These factors include aesthetic quality together with actual and potential recreational and amenity use of the environment. Champ *et al.* (1981) and Tidmarsh *et al.* (1986) highlighted the potential for the formation of surface slicks and the presence of offal on beaches. Similarly, large amounts of shell could be washed up on an otherwise sandy, public amenity beach, if dumped inappropriately. Care must be taken to avoid this. Aesthetic considerations will also include the attraction of undesirable numbers of seabirds which, in areas used by the public, not only cause a nuisance but also pose a threat to the quality of bathing waters and drinking water supplies due to the presence of faecal coliforms in their droppings.

**Stage 3. Disposal site characterisation:**

This step takes place in two phases: a desk study in which sites are investigated from published and unpublished literature, navigational information and local knowledge. At this stage several potential sites could be chosen which would then be further investigated. A thorough desk-study may provide a large amount of the information required in the assessment and thus will provide a cost-effective assessment and an appropriate choice of site. In cases where insufficient information is available, as is often the case, additional information will have to be obtained to carry out the thorough impact assessment; the latter will always be required and may or may not require additional field data. The latter may be required to produce the baseline data against which future monitoring and the detection of change will be carried out.

Characterisation of the disposal site should include measurements of all components which will be influenced by waste disposal. That is, the water surface, water column, sediment-water interface and the sediment itself. While much of the physical information can be obtained from Admiralty Charts, local fishermen's knowledge of the sites should also be obtained since they will be familiar with tidal and other conditions. Similarly, before a field survey of the physical, chemical and biological features of the seabed is carried out, it is necessary to review other surveys which may have been carried out in the proposed dumping area. Such surveys may include biotope surveys carried out by a combination of acoustic, photographic and faunal sampling techniques, including infaunal grab sampling and epibenthic trawl sampling; chemical data for the area may also be available. This information may be obtained from the fisheries advisory organisations (CEFAS, FRS, DARD(NI)), the environment protection agencies (the Environment Agency, SEPA, EHS), the nature conservation bodies (English Nature, SNH, CCW, JNCC), academic organisations and commercial research institutions (subject to approval by the client for whom the survey was carried out). A review of the available data will highlight the presence of any sensitive habitat or community types, together with the potential for waste accumulation and associated environmental impacts. It is emphasised that field surveys are potentially expensive and so all sites unsuitable for disposal require to be excluded sufficiently early in the process.

a) Hydrographic information

Admiralty Charts should be used to provide geographical, topographical/bathymetric and hydrographical information. This includes the location, shape, size and distance from land (km from MHWS) of the proposed site, the water body type (open sea, coastal, estuarine), depth, the direction and strength of the residual currents and the wave strength in the area. Data on the strength and direction of the prevailing winds should also be obtained and used in combination with the hydrographic data in order to make an assessment of the likelihood for dispersal (i.e., is the proposed site in a high or low energy area).

The topographic/bathymetric features of the receiving environment require characterisation (e.g. using acoustic techniques such as side-scan sonar, information from admiralty charts, previous surveys and photography) since changes to these properties can impact upon the near bed flow conditions and, in turn, impact upon the biological communities in the area.

b) Water quality

Determination of the water quality in the area will also be necessary, including depth profile measurements of dissolved oxygen, turbidity and salinity with near bed (e.g., within 1 m of the bed) measurements of DO and turbidity being particularly important since this is likely to be the depth range to show the greatest impact. Under certain circumstances it may be considered necessary to consider the following parameters:

1. Given that seafood processing waste is unlikely to contain high concentrations of contaminants (in comparison, for example, to dredged material), the analysis of water concentrations of persistent contaminants may not necessarily be required. However, if there is potential for the liberation of contaminants from the sea bed, for example as a result of changes in oxygen availability, or where mildly contaminated waste is to be dumped in a concentrated area, water quality analysis may be necessary.

2. Determination of suspended particulate organic carbon may be necessary if a large proportion of the waste is organic. Methods for its determination include Loss on Ignition (%LOI), wet oxidation techniques or the use of a CHN analyser. The Biological Oxygen Demand of the material being dumped would also indicate the potential for decreasing water oxygen concentrations; the latter will also vary with season, thus giving a temporal variability to the magnitude of effects.

However both these requirements can be eliminated if a sufficiently dispersive site can be chosen in the initial desk study.

c) Sediment quality

Characterisation of the sea bed at the proposed dump site requires physical, chemical and biological measurements. These include characterisation of the sediment type (e.g. rock, sand, mud) and the particle size distribution, together with the organic content and redox (reduction-oxidation potential) conditions. The latter is especially important in the case of a high organic discharge with the potential to cause anoxic conditions. These physico-chemical properties, together with hydrographic information will give an indication of sediment behaviour in the receiving area, allowing classification into the following types of environment:

- accreting, soft sediment (mud/muddy sand);
- moderately accreting, subtidal sand banks;
- moderately dispersing subtidal sand and gravel banks;
- highly dispersive area composed of a gravel, shell and mineral bed.

Exposed bare rock is unlikely to show any adverse effects although the waste can accumulate in pockets, crevices, etc thus creating localised anoxic areas. Biological investigations should include determination of the infaunal benthic community structure (and/or a biotope description), the epibenthos, demersal fish and determination of the main feeding types present, such as suspension feeders, deposit feeders. These determinations can be carried out by grab and core sampling and video observation.

It may also be necessary to determine the concentrations of chemical contaminants within the sediments since, following the dumping of organic waste, changes in the aerobic-anaerobic balance may lead to the liberation of these substances causing contamination of the water column. However, as above, sites which are already considered this contaminated should be avoided.

**Stage 4. Suitability of the proposed site for waste disposal.**

In cases where inert particulate material is to be dumped, such as shell material, once both the waste and environmental characteristics have been defined, an assessment of the similarity between the waste material and the physical properties of the sea bed must be made to determine the suitability of the site for waste disposal. That is, if the particle size of the waste is similar to that at the receiving sediment then it would not alter the ambient sediment conditions. This may not be possible in all cases although, depending upon the potential for dispersion, may not necessarily be cause for concern. MEMG (2003) advised that the potential for concern where the waste type and the sediment type were dissimilar should be categorised in terms of low, medium or high.

**Stage 5. Potential for dispersal (spatial impacts).**

Due to tidal and wave generated currents, it is unlikely that all of the waste will remain permanently where it was initially dumped, regardless of whether the area is classed as dispersive or low energy. There is always potential for some dispersal of the material, the degree being dependent on the nature of the waste (particle size and buoyancy) and the strength of the prevailing currents in the area. The disposal sites, together with the immediate area are termed 'near-field' sites. In high energy areas, where waste may become widely distributed, the areas which may potentially be affected are termed 'far-field sites'. An assessment of the degree of dispersal is required in order to identify the far-field sites and to assess the potential impact and monitoring requirements at both near and far-field. Estimations of the degree and direction of dispersal can be made from the Admiralty Charts. In areas where there is less information available, the collection of hydrographic information (e.g. using drogues, depth reading current meters (DCRM), acoustic doppler profile imaging (ADCP)) , together with modelling studies, may be required to give a more accurate indication of the fate and thus the effects of the dumped material.

**Stage 6. Waste residence time (temporal impacts).**

Based on the nature of the waste, the disposal method, the dispersive characteristics of the receiving environment and the potential for the removal of waste by scavengers, an estimation of duration of any impact must be made. That is, the length of time the waste is expected to remain on the surface, in the water column or on the seabed should be stated as short, medium or long term. As indicated in the case-studies, the length of time taken for dispersal and degradation of newly deposited waste may be difficult to predict. For example, Ramsay *et al.* (1998) found that the presence of fishery discards did not necessarily attract scavengers if food availability was naturally high. Stevens & Haaga (1992) reported rapid decomposition and consumption of waste (composed of fish heads, bones and viscera) in Kodiak Bay, Alaska and reported a half life of 8.7 days with no waste remaining after 33 days. In this case, waste was dumped over a widespread area. In contrast, Tetra-Tech found that waste dumped in piles (i.e. concentrated or aggregative dumping) was still present, although greatly reduced in volume, after eight months.

**Stage 7. Potential environmental impact.**

The potential for environmental impacts at near-field and, depending upon the expected degree of dispersal, far-field sites must be assessed, taking into consideration the likely timescale of the impact. Where small quantities of uncontaminated waste (as in the case of seafood processing waste), are dumped the potential for concern may be low, particularly if the waste is composed predominantly of inorganic material. However, the disposal of large amounts of organic material at a single site will give rise to greater potential for concern due to the effects outlined in Section 4.1. Furthermore, dumping coarse particulate material in a low energy, muddy area will also lead to cause for concern due to potential changes in the ambient sediment properties and subsequent effects on the biological communities. In the case of far-field sites, an assessment of the potential impacts and full characterisation of the site properties (physical, chemical, biological features and socio-economic interests, as outlined in stages 3 and 2, respectively) is only necessary where there is a realistic potential for impact.

As in the initial site characterisation process, the potential impacts should be assessed at all vertical levels throughout the system including the water surface, water column, sediment water interface and the sediment. At each level, the physical, chemical, biological (as outlined in stage 3), aesthetic and socio-economic impacts (stage 2) should be considered. In addition to the factors outlined above, an assessment of the surface water quality (e.g. by photography or aerial photography for widespread effects) must be carried out to ensure that there is no long term development of a slick.

**Stage 8. Requirement for monitoring**

As stated in Section 6.1, the purpose of monitoring is to demonstrate compliance with the licence conditions and/or verify the impact assessment (Table 4) (MEMG, 2003: London Convention, undated). The actual, potential or perceived effects of waste disposal will dictate the nature of the monitoring required although the components of the monitoring programme must relate to the cause for concern (MEMG, 2003). That is, the monitoring requirement will be based on the identification of features (physical, chemical, biological, socio-economic) which could potentially be affected, together with the timescale of the dumping operation and the expected volume/type and behaviour of the waste. The frequency of the monitoring will depend upon the site characteristics (near and far-field) and the expected level and timescale of the impact. Monitoring requirements are therefore site specific (OSPAR, 1998).

**Table 4. Generic guidelines for monitoring disposal sites (from MEMG, 2003).** (LC – London Convention; other abbreviations as in the text)

| | |
|---|---|
| 1. Overall aim | Ensure no unacceptable loss of quality or deterioration to the health of a system in its structure or functioning nor hindrance to the uses and users of an area. |
| 2. LC and OSPAR guidelines adopted | Demonstration of compliance with permit conditions and that changes in the condition of the receiving area are within those predicted in the Impact Hypotheses. |
| 3. Objectives defined | Environmental and Ecological Quality Objectives (EQO/EcoQO) would be defined as Null Hypotheses and incorporated into generic indicators of favourable conditions at each site. |
| 4. Standards adopted | Any available and accepted Environmental and Ecological Quality Standards (EQS/EcoQS) for waters and sediments will be used in monitoring. |
| 5. Monitoring strategy | Use of BACI approach to determine spatial and temporal changes in environmental quality. |
| 6. Action Point and feedback monitoring | If monitoring suggests unacceptable effects, the dumping operation must cease, pending further assessment. If un-anticipated effects are detected, such evidence should act as a trigger for further investigation. |
| 7. Audit of monitoring. | The methods used are subject to Best Available Practice and AQC/QA (Analytical Quality Control/Quality Assurance) and scoping documents and reports are independently peer reviewed. |

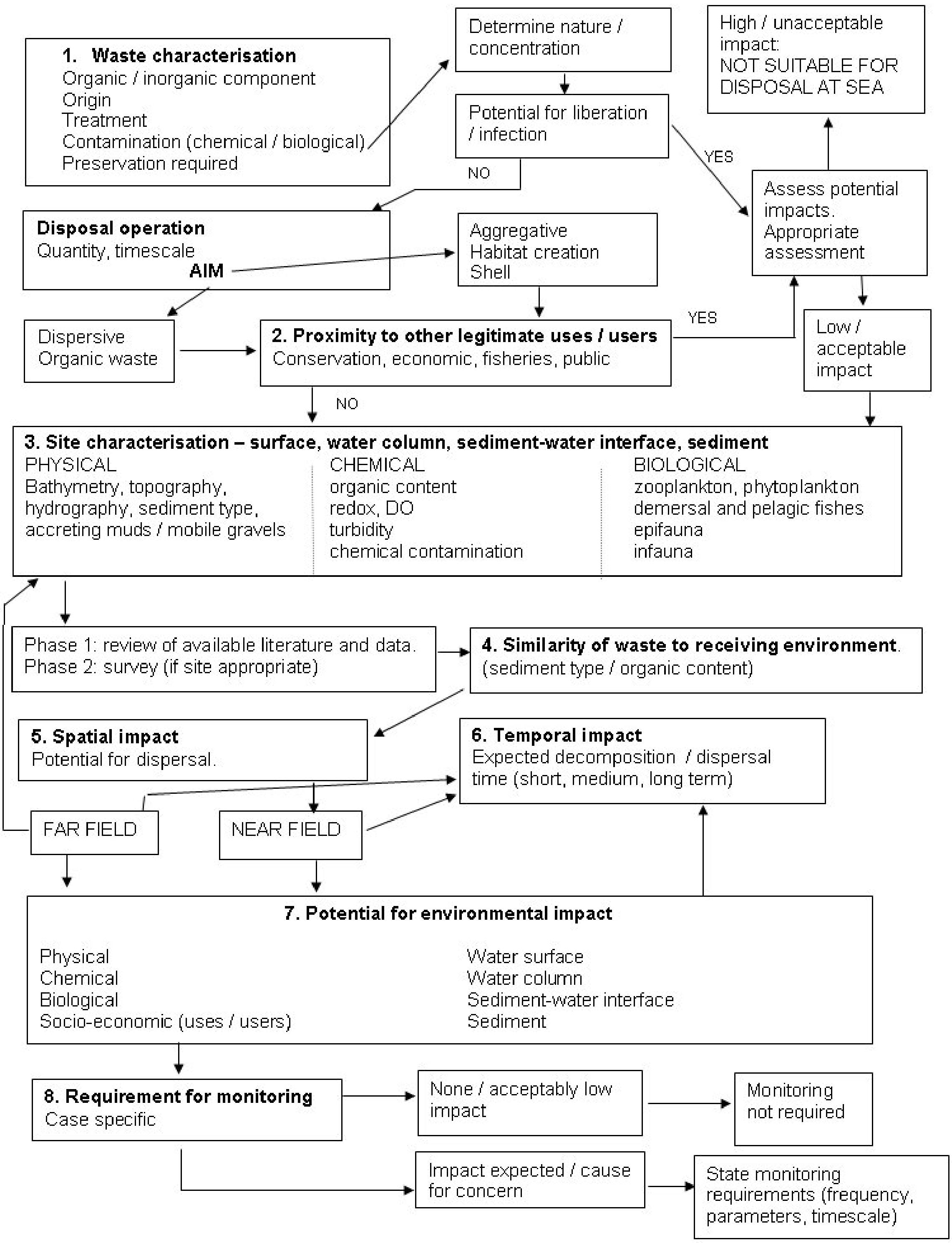


**Figure 8. Guideline to disposal site selection (adapted from MEMG, 2003).**

# 7. DISCUSSION

## Environmental consequences of sea-disposal of fish and shellfish waste

The potential effects of disposal can be regarded as a set of bottom-up causes and primary effects, in which the physical and chemical systems, both in the water column and on the bed, are altered and which in turn affect the health of the biological system. The eventual effects on the higher levels of the biological system and its uses by Man can be regarded as a set of high level responses, e.g. the effects on the higher levels of the ecological system (such as fishes, seabirds and marine mammals) as well as on fisheries and conservation objectives (Elliott *et al.,* 1998).

The effects of organic waste on the marine environment and the effects of waste disposal at sea (e.g. sewage, dredged material) are well documented (Pearson & Rosenberg, 1978; MAFF, 1993; Diaz & Rosenberg, 1995; MEMG, 2003; Elliott & McLusky, 2004). As shown here, depending upon the size, shape and buoyancy of the material, the disposal of organic and mixed material will have the potential to affect the water column, the bed conditions and their biota (Figures 5 and 6). Reductions in water clarity through an increase in turbidity could affect the primary production by the phytoplankton. The release of any organic material may result in changes to the chemical environment, potentially leading to anoxia in the sediments and bottom waters and the release of toxic materials (methane, ammonia and hydrogen sulphide). Furthermore, chemical changes in the environment can lead to the liberation of toxic substances sequestered within the sediments or contained within the waste (Libes, 1992).

The deposited material will change the nature of the bed sediment, and if it is of a different particle size, it can have a smothering effect on the bed community. Both of these features will affect the structure of the bed community and, in turn, the demersal fishes feeding on that bed community. The disposal of solid (shell) waste will alter the bed topography and bathymetry which will influence the substratum as well as changing the overall hydrodynamic regime of the area. The structure and functioning of those bed sediments and the overlying hydrographic regime (water currents, tidal circulation, etc.) will be intimately linked to the structure and functioning of the bed biological community, principally the invertebrates (Elliott *et al*., 1998). In turn this will influence the fishes and, in nearshore areas, the birds feeding on those invertebrates. Ultimately, all of these effects have the potential to influence the fisheries and nature conservation value of the area (Figures 5 and 6).

Whilst the disposal of seafood processing waste bears some similarities to the disposal of both inert and organically-contaminated dredged material and sewage sludge, the quantities are much less (MEMG, 2003; Large, 2004). MAFF (1993) stated that, in many cases, disposal at sea is the Best Practicable Environmental Option which, if carried out appropriately, does not cause unacceptable environmental impacts (Heap *et al.,* 1991; MEMG, 2003). Furthermore, seafood processing waste on the seabed will attract epibenthic scavengers in a similar manner to that of discards from trawlers (Ramsay *et al*., 1998; Veale *et al.,* 2000). Such species not only respond quickly to the increased food supply but also rapidly remove it from the seabed so that the impacts associated with the disposal of organic waste may be minimised and short lived. However, it should be noted that scavengers do not always respond in this way, particularly in areas where the food supply is already plentiful (Ramsay *et al*., 1998). The dumping of shell material also has potential environmental benefits such as habitat creation and a positive impact on epifaunal species diversity (Guay & Himmelman, 2004). Other potential beneficial uses of shell material include protection against saltmarsh erosion (Meyer *et al.*, 1997).

As indicated above, there are many potential impacts of the disposal of the waste but those impacts will only occur if there is a poor site selection. As shown in the present study, although there is little available information regarding the specific effects of the disposal of seafood processing waste at sea, some of the case studies have found the impacts to be negligible. Stevens & Haaga (1992) found that, if material was dumped over a large area (i.e., dispersive dumping from a moving barge), waste accumulation did not occur in the long term and the decay/removal rate of the waste was rapid (half life of 8.7 day) with no waste remaining after 33 days. It should be noted that the concentration of fish bones and frames was higher at the disposal site than at the control site. No impacts indicative of faunal community stress were recorded but it should be noted that faunal sampling was restricted to the epibenthic community and no information regarding the effects on the infaunal community was given. Furthermore, the dumping of sewage sludge at sea has taken place in the past in UK waters with no obvious direct effects on the benthic communities present (Heap *et al.*, 1991; MAFF, 1993). It is of note that the addition of organic material to the seabed in some areas actually enhanced the benthic community which, prior to waste disposal, was impoverished as a result of strong currents and low levels of organic material (MAFF, 1993).

However, large scale waste accumulation and the associated effects on the seabed were noted following the disposal of seafood processing waste in King Cove, Alaska (USEPA, 1998). Modelling was carried out by the USEPA (1998) in order to determine the total maximum daily load which could be dumped without resulting in waste accumulation. This was based on various particle sizes and densities of waste, together with the settling velocity and current required for resuspension following settlement. Whilst the fate of the waste in the sea will be site specific according to local circulation patterns and hydrographic conditions, such factors are need to be considered when selecting disposal sites. Similarly, waste was still present eight months after dumping in the Tongass Narrows, Ketchikan, Alaska (Tetra Tech, 2004). However, in this case, the waste was deliberately dumped in piles and it is suggested that dispersive dumping would greatly increase the rate of decomposition, dispersal and removal from the sea bed.

If the disposal of seafood processing waste at sea is to be carried out, it is important that an appropriate site is chosen and that sufficient baseline information is collected to ensure that the site is able to receive a defined volume and type of waste without resulting in any unacceptable impacts (Figure 8). The latter implies that a site is within its assimilative capacity. The likely impacts should be defined, considering the physical, chemical, biological, and socio-economic aspects at all levels within the system (water surface, water column, sediment-water interface, sediment). The potential for waste dispersal must be considered together with the potential to impact upon far-field as well as near-field sites.

It is of prime importance that the characteristics of the waste match the characteristics of the sediment in the receiving environment. Ideally, organic waste should be dumped using a highly dispersive method and/or placed in a highly dispersive environment, to increase the rate of degradation (Stevens & Haaga, 1992; Tetra-Tech, 2004) and availability to scavengers. This would reduce the magnitude and timescale of any impacts. Conversely, if the aim is habitat creation using shell, the material should be dumped over a smaller area to gain maximum benefit.

As indicated earlier, the challenge for marine environmental managers is to achieve a sustainable use of the marine environment. That sustainable solution will occur if it fulfils the 6 tenets in being environmentally sustainable, technologically feasible, economically viable, socially desirable and/or tolerable, legislatively permissible and administratively achievable. In this discussion, these require to be revisited and summarised in the light of the present report.

## Environmentally sustainable

The methodology and procedures for assessing marine environmental impacts are well-defined and, in the case of seafood-waste, can be based on the approaches to the disposal of sewage sludge, other organic wastes and both inert and organically contaminated dredged material. Once the nature of the material to be disposed and the potential receiving area have been characterised then the fate and the effects of the disposal can be determined.

The characterisation will require an initial desk study and may require, as licence conditions, a pre-disposal field survey and monitoring over a period of time. An initial assessment will ensure that a site is chosen where there are likely to be minimal environmental impacts, especially where highly dispersive sites and a dispersive method of disposal can be achieved. However, there may be circumstances such as the disposal of shell waste where it is necessary to match the substratum on the seabed or use the shell as a substratum for habitat enhancement. This implies aiming to dispose of the shell on specific habitats, thus constituting a beneficial use of the waste.

The present study has indicated that, whilst the disposal of seafood processing waste has been carried out in many areas, there have been very few quantitative impact studies. Similarly, there have been some impact studies of the disposal of other, broadly similar, types of waste at sea, but there is a lack of case studies relating specifically to the disposal of seafood waste. Before a comprehensive assessment of the impacts of such disposal can be carried out, more quantitative data are required together with testing and validation of the conceptual models presented here. In addition, more details of the waste characteristics are required in order to determine the environmental fate and potential risks in different environments and numerical modelling of the waste may be required to indicate areas at risk.

## Technologically feasible

In order for successful environmental management of the sea disposal of seafood waste, there are several technological aspects which need to be considered. Most importantly, the method of handling the waste will influence the potential environmental effects. As indicated in the case-studies discussed here, maceration could be used to give an optimal size for dispersion, degradation and assimilation; similarly, hygienic storage and transportation (e.g. freezing) could be used to reduce the hazard to health, nuisance and to prevent the waste becoming anoxic during storage. The waste may require to be containerised to reduce transport hazards.

Vessel technology will be required to ensure adequate dispersion of organic waste, from simple man-handling of the waste overboard to pumped systems used while the vessel is moving as this will ensure widespread dispersal. Dumping large quantities in one place is a less favoured environmental option although for the disposal of shell on a specific substratum, more specific targeting may be required.

Although outwith the remit of the present report, a full technological appraisal needs to be performed.

## Economically viable

Under the internationally and nationally-adopted polluter-pays philosophy, the costs of treatment, licensing, movement and disposal of waste, compliance with any licence conditions, and monitoring should be borne by the waste producer. The main costs associated with sea disposal are:

1. The annual licence fee payable to the Government for sea disposal, this may differ between regulators.
2. There is the cost of the initial site selection, environmental impact assessment and monitoring. These will include the consultation, desk-based and field studies and may require the costs of testing for contaminants.
3. There are costs attached to storage and transport, by land and sea to the disposal site, including any costs of additional treatment such as freezing and maceration.

There is the need to rigorously assess the above costs, to compare the costs of sea-disposal with those of other methods, such as landfill, and to include all energy costs for each option. Consultation with industry will be required to determine an economically viable solution although, as indicated in the present report, the economies of volume are difficult to achieve with many types and sources of fish and shellfish waste.

There will be the need for industry to make initial investment in site selection, to cover the costs of impact assessment and monitoring. The monitoring can be cost-effective as long as a thorough desk-based assessment is carried out and appropriate methods are used. The periodic monitoring of disposal sites may be necessary, according to licence conditions, but this will also provide valuable information for future licence applications. The provision of further information and data may simplify the process for future applications and possibly reduce costs. However, as it is possible that 3-year licences are issued, then a re-assessment will be necessary at each licence renewal taking into account alternative means of disposal, the results of any monitoring and the views of consultees. For a major disposal operation, a full EIA could be required in the future.

## Socially desirable or tolerable

The Best Practicable Environmental Option and Environmental Impact Assessment procedures include elements of social acceptability, through stakeholder consultation, and the determination of amenity impact. Hence these will incorporate and address any social objections to this waste disposal process. The BPEO procedure is required to be used for the whole product cycle and as a rigorous assessment of all alternatives, both land and sea-based. While the disposal to sea of industrial and sewage waste is regarded as socially unacceptable and was therefore stopped, the acceptability of returning and the direct recycling of matter that has been removed from the sea require to be assessed.

Furthermore, if seabed habitat enhancement using shell is a viable option then this may be regarded, in present thinking, as a beneficial use of the material.

### Legislatively permissible

The activity is permitted under international conventions and UK legislation as a licensed operation. However the present licensing approach requires to be clarified and anomalies need to be removed. For example, discussions for the present project could not determine whether, or if so why, this disposal came under the Animal By-products Act(s). There is a potential anomaly that whereas fish and shellfish discard and bycatch material taken on board a vessel during commercial fishing can be returned to the sea without licence even if it produces anoxic areas of the seabed, that taken into an onshore processing plant and then discharged back into the sea requires a licence. Similarly, if fish and shellfish impinged in cooling water intakes for power stations are immediately returned to sea then this is deemed to be an organic discharge needing consent from the environmental protection agencies. Under the control of pollution legislation, the operators in this case are deemed to be causing pollution whereas discards from vessels during fishing are not. It is possible that these anomalies will be discussed in the proposed Marine Bill.

### Administratively achievable

The licensing bodies, Defra, DoE(NI) and SEERAD and their advisors (CEFAS, EHS and FRS) implement the licensing scheme in force under the Food and Environmental Protection Act II which enables the granting of licences for dumping of seafood factory waste at sea. At present this is done on an *ad hoc* basis so that there is the potential for a more coordinated approach. For example, the licensing of several vessels to dispose of seafood wastes at the same site would improve economic viability as long as the assimilative capacity of the site is sufficient.

# 8. CONCLUSIONS

The sea-disposal of fish and shellfish processing waste is concluded to be a viable option which can be accomplished in a sustainable way and which the six tenets of environmental management, described earlier, can be satisfied (see Text Box below).

| Tenet of sustainable marine management: | Achieved: |
|---|---|
| Environmentally sustainable | as long as there is an adequate waste characterisation and site selection procedure |
| Technologically feasible | as long as the methods for suitable placement are devised |
| Economically viable | with economies of scale and a cost-benefit assessment |
| Socially desirable/tolerable | following agreement by stakeholders |
| Legislatively permissible | at a basic level but there is the need for clarification |
| Administratively achievable | as the statutory bodies and their advisors are in place |

That environmental sustainability is dependent on a satisfactory site selection and an appropriate site can be selected given a well-defined set of aims and the means to achieve those aims (see Text Box below).

| Aim to: | Achieved by: |
|---|---|
| minimise interference with uses and users | desk-study and consultation |
| minimise the environmental impact | desk-study and fieldwork |
| evaluate options for disposal | desk-study and consultation |
| determine the capacity of the disposal site | desk-study, modelling and fieldwork |
| characterise the receiving environment | desk-study and fieldwork |
| determine the transport of material thus influencing near and far field effects | desk-study and modelling with field validation |
| determine the accumulating/dispersing nature of the site | desk-study and fieldwork |
| consider the acceptability of any effects | desk-study and consultation |

As shown here, the sea-disposal of fish and shellfish waste requires a concerted assessment and further discussion between all parties. There is the need for a collaborative approach between the industry and the regulators with input from scientific, technical and economic expertise. This would indicate the way ahead to minimise or prevent problems.

There is sufficient knowledge of marine processes and the assimilation of this type of waste to conclude that environmental impacts will not occur if the waste is disposed of in an appropriate manner. The use of the Best Practical Environmental Option procedure and by carrying out a sufficiently rigorous Environmental Impact Assessment will ensure that the above 6 tenets of sustainable environmental management are fulfilled and that all stakeholders are agreeable to the solution adopted.

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