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A REVIEW OF THE ECOLOGICAL IMPLICATIONS OF MARICULTURE AND INTERTIDAL HARVESTING IN IRELAND

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INTRODUCTION

In Ireland, mariculture activities are increasing significantly every year and there is some concern over the sustainability of these activities. The purpose of this report is to provide comprehensive, up-to-date information on the ecological impacts, of all marine species cultivated or harvested intertidally in Ireland, on a species by species basis. The report will be limited to marine species currently cultivated/harvested, and methods currently practised in Ireland, and will include those likely to be adopted by the year 2001. In addition, this report will recommend management measures to counteract any negative ecological effects. This information will help managers to make decisions on licence applications for aquaculture and enable them to point existing operations towards sustainability.

Unlike a laboratory experiment we cannot control, or even know, all the factors or processes involved and the purpose of this report is to provide indications of likely impacts. What is proven true at one site may not be valid at another. One difficulty with this type of report is that by listing all the impacts associated with cultivation of a species the world over a false impression may be created. The impacts are as variable as the sites.

This report only references literature from the Atlantic coast of Europe (Norway to Iberia), the Atlantic coast of Canada/USA and from other temperate seas of the world. As stated above, only the species cultured or harvested intertidally in Ireland are included. An exception to this, are some rare, key papers on a closely related species which have been included to make a point which may otherwise have been missed.

MATERIALS AND METHODS

An exhaustive literature search was carried out on the Aquatic Science and Fisheries Abstracts database (1988-1996) using key words (Appendix 1 and 2). The same keywords were used to double check that no reference had been missed on the Aquaculture and the Environment bibliography database on the internet. There appeared to be a dearth of information under some headings. These headings were checked again on the CD-ROM (Science Citation Index (January1992-May 1996)) at Trinity College Dublin, but very few additional papers were located. An additional source was the BioMar database at Trinity College. This database was of particular value as a lot of 'grey' literature was listed. The reports identified as being of interest were then requested from the relevant bodies.

Various bodies, and a few individuals, were contacted to request relevant unpublished material. Bord Iascaigh Mhara (BIM) supplied information on which species are being cultivated at present and which ones are likely to be cultivated in the near future. The Department of the Marine supplied information on methods used for intertidal harvesting in Ireland.

In order to produce the most comprehensive report possible given the time restraints, it was agreed to limit, where possible, literature to post 1991. Pre 1991 literature which was included consisted primarily of key references or was used where more up-to-date information is not available.

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SECTION 1

FINFISH CULTIVATION

ATLANTIC SALMON (*SALMO SALAR*)

1.1 Salmon cultivation in Ireland

Ireland is one of the world leaders in Atlantic salmon production. In 1995 production was 11,811 tonnes. The natural breeding cycle of the salmon requires spawning in freshwater, migration and growth in the sea until the sexual maturity is reached, at which point the fish returns to its parent river to spawn. This anadromous habit causes salmon farming to be divided into two separate stages, freshwater and marine, for which clean, cool, well oxygenated water is required. The complete life cycle of the salmon can be reproduced in a period of 3-4 years using intensive farming systems (E. C., 1995).

The production process begins the freshwater hatchery. The fish are then transferrred to freshwater tanks or cages and here the fish mature from 'fry' to 'parr' and then to 'smolt' which is the transitional period of adaptation to sea water. After smolting, fish (aged about 17 months) are transferred into, floating net cages at sea, where market size (ranging from 2-5 kg) is obtained in a period of 1-2 years (E.C., 1995).

1.2 Data available

There is a vast quantity of data available on salmon farming and the body of literature increases every year. Given that Ireland is now a world leader in terms of production, it is surprising that so few published papers are based on studies carried out here. If we wish to develop a sustainable industry, research should keep pace with production. The conditions in Ireland may, or may not, be comparable to those in other countries.

1.3 The ecological impacts of salmon cultivation

The ecological impacts of salmon farming will be discussed under the following headings

1.3.1 Waste
1.3.2 Benthic organisms
1.3.3 Nutrient balances
1.3.4 Escapees
1.3.5 Disease
1.3.6 Chemicals

1.3.7 Predators

1.3.1 Waste

Organic waste is known to accumulate under fish farms cage sites. It consists mainly of waste feed and faecal pellets (O'Connor *et al*., 1993; Scott *et al.*, 1995) and has a high nitrogen and phosphorous content (Kupka-Hansen, 1994; Ackefors and Enell, 1990). Rates of waste accumulation show large annual variations as they mainly follow the feeding strategy with high outputs in the summer and in the autumn (Hall *et al*., 1990). The amount that accumulates will depend on currents, sedimentation rates, decomposition rates, the abundance of epi- and infauna and the rate of resuspension of the sediment (Holmer and Kristenson, 1992; Findlay and Watling, 1995).

Waste accumulation of 18 cm has been found, in Western Sweden, by measuring the sediment depth profiles of carbon and phosphorous (Hall *et al*., 1990; Hall and Holby, 1991). In other studies, from 1-40 cm of accumulated waste has been recorded (Braaten *et al*., 1983; Kupka-Hansen *et al*., 1991). The organic wastes per tonne of fish produced are presently estimated to reach 2500 kg wet weight per tonne of fish live weight (Ackefors and Enell, 1994). Feeds have improved in recent years and this has greatly reduced overall environmental loads, which are presently calculated as 10 kg of phosphorous and 60 kg of nitrogen per tonne of fish produced (Ackefors and Enell, 1994). Gowen and Bradbury (1987) estimated that the deposition of organic waste beneath a fish farm might be as high as 10 kg. $m^{-2}.yr^{-1}$ directly beneath the cages and 3 kg. $m^{-2}.yr^{-1}$ in the immediate vicinity of the farm. In contrast, some farms do not show any accumulation of waste, this is probably due to scouring by water currents at this site (Findlay and Watling, 1995).

Where waste accumulates, the microorganisms in the sediment work to oxidise and breakdown the waste. Krost *et al.* (1994) observed that the sediment under the farm was organically enriched 1.5 to 3.5 fold above the control, and the rates of decay of organic carbon and oxygen uptake were high and recorded as 100-150 mmol m^{-2}. d^{-1}. In breaking down the waste, the available oxygen is used (Brown *et al*., 1987; Gowen *et al.,* 1988) and so, the redox potential becomes negative and the sediments become anoxic (Krost *et al*., 1994).

Negative redox potentials in the sediments under fish farms have been noted by many authors (Hall *et al*., 1990; Wildish *et al*., 1990; Hargrave *et al*., 1993; Krost *et al.*, 1993). However, this has not been observed at all sites. Gowen (1990) noted that In Kilkieran and Bertraghboy Bays, Co. Galway Ireland, the surface redox potential was positive, out-gassing was limited to directly beneath the cages and there were only small increases in the level of sedimentary organic carbon under the cages compared to control stations.

Another characteristic stage noted by authors from Canada to the west of Ireland, is the colonisation of the sediment by the sulphur bacteria *Beggiatoa* spp. (Brown *et al*., 1987; Hall *et al*., 1990; Krost *et al*., 1994; O'Connor *et al*., 1993; Findlay and Watling, 1995). These bacteria break down the organic wastes anaerobically via fermentation and sulphate reduction with concomitant gas production. Samuelsen *et al*. (1988) analysed the sediment gas and found that it consisted of methane (70-90%), carbon dioxide (10-30%) and hydrogen sulphide (1-2%). This was also observed by Wildish *et al.* (1990) and Gowen (1988). Hargrave *et al.* (1993) warned that concentrations of >100 mM sulphide in sediment pore water could be toxic to benthic fauna.

Krost *et al.* (1994) noted that the reduced conditions under the Kiel fjord fish farm, in Norway, rendered improbable any nitrogen release with a higher oxidation state than NH_4. Practically all of the mineralisation was due to sulphate reduction. Indeed, Kaspar *et al.* (1988) failed to detect in-situ nitrification in the sediment directly under a salmon farm. Krost *et al.* (1993) noted complete disappearance of hydrogen sulphide in the pore water only a few metres from the grossly polluted areas under the fish cages which is a remarkable proof, in this case, of the limited area affected by organic matter.

In order to quantify the magnitude of the effect of any fish farm, Gowen and Bradbury (1987) developed a model which gives an estimate of the spread of organic waste and carbon input to the sediment beneath and adjacent to the farm cage. The results of Brown *et al.* (1987) conform well to the values predicted by this model. This provides a means of assessing the minimum distance that a farm should be sited from other activities which use the seabed. This model has since been updated and refined by other authors to provide a more realistic model for the impacts of organic enrichment from marine aquaculture (Hargrave, 1994).

1.3.2 Benthic organisms

The enrichment of sediments and resultant anoxia often recorded under the cages has been shown to result in changes in macrofauna (Gowen, 1990). Macrofauna are defined as animals retained on a 1 mm sieve which live on and in the sediment (Gowen, 1990). Every site is different (Findlay and Watling, 1995) and the rate of impact and recovery will depend on the hydrographic condition and farm management (Gowen *et al.*, 1988).

In Norway, seven comparable fish farm sites and a control station were investigated four times in one year. It was found that when the thickness of the accumulated waste exceeded 20 cm, the larger fauna (i.e.>5 mm) disappeared, and the decomposition rate of the material decreased (Kupka-Hansen *et al.*, 1991). Nearly all authors suggest that there is some sort of zonation occurring beneath and surrounding fish farms. There are many theories on the zonation and many authors have defined characteristics of the zones and indeed their size for their own study sites. These zones are usually based on biological impact as the infauna are sensitive to changes in the sediment which are not physically or chemically detectable, and thus the zone of impact is often larger (Weston, 1990) .

Brown *et al.* (1987) observed that benthic fauna showed marked changes in species number, species diversity, faunal abundance, and biomass in the region of the fish farm and identified four zones of effect:

Zone 1 - An azoic zone directly beneath and up to the edge of the cages.
Zone 2 - A highly enriched zone dominated by *Capitella capitata* and *Scolelepis fuliginosa* from the edge of the cages out to about 8 m.
Zone 3 - A slightly enriched "transitional" zone occurred at ^{2}25m,
Zone 4 - A "clean" zone > 25 m. At 120 m out from the farm conditions could be considered normal.

Many authors found *Capitella capitata,* a species known for its tolerance to anoxic conditions and high organic enrichment, either directly beneath the farm (Pocklington *et al*., 1994) or close to the farm (Gowen *et al.*, 1988; Weston 1990; Lim, 1991). Lim (1991) observed that 55 m away from the salmon farm species diversity and biomass peaked. Over the sampling period biomass levelled off at 185 m from the farm (Lim, 1991). This could be considered to be equivalent to zone 4 using the zonation suggested by Brown *et al.* (1987). Weston (1990) reported that benthic community effects were apparent to a distance of at least 150 m. Johannessen *et al.* (1994) reported no effects at a distance of 250 m from the farm.

Henderson and Ross (1995) investigated the use of macrobenthic infaunal communities for monitoring the impact of marine cage fish farming. Their sites were all sheltered sea lochs. They found that macrobenthic infaunal responses, though not fully understood, provided the best measure to date of determining the impacts of organic wastes from cage fish farming and a possible way forward in developing benthic environmental quality standards for aquaculture. Like Findlay and Watling (1995), they emphasised the importance of site individuality, and hence the difficulties of setting environmental quality standards across the industry. Henderson and Ross (1995) proposed a zonation system of impacts, using faunal characteristics and indicators for light to moderate organic enrichment. They highlighted the difficulty in separating impacted sites from background enhancements. While areas close to their study cages were grossly impacted, low species diversity, high abundance and perturbed dominance of opportunist species also occurred up to 25-30 m from occupied cages and in exceptional circumstances, even greater distances were affected (Henderson and Ross, 1995). This is in general agreement with other authors. However, the size of zones vary according to site.

Most research on the impact of salmon farming has been carried out in Scottish sea lochs or Norwegian fjords. These sites tend to be more sheltered and less well flushed than off-shore sites. Frid and Mercer (1989) noted that although fish farm developments were initially located in sheltered environments, development is now occurring in areas with strong tidal flows. This is certainly the case in Ireland. Significant benthic enrichment is known to occur at sheltered sites, but the effect is highly localised. The siting of caged fish farms in macrotidal environments may reduce the environmental impact of the industry on benthic communities at the farm site. However, there exists the potential for accumulation of farm wastes in nearby sedimentary sinks (Frid and Mercer, 1989). The longer wastes are resident in the water column (especially in regions with already high nutrient loadings and, or long flushing times) the greater the potential, of such wastes, to lead to the stimulation of phytoplankton blooms (Frid and Mercer, 1989.).

In 1990, the Department of the Marine in Ireland commissioned a report on Irish marine salmon sites. In Kilkieran and Bertraghboy Bays the zone of enrichment was not considered to have extended beyond a distance of 10 m. Gowen (1990) concludes that in general the level of enrichment which has taken place in the vicinity of fish farms in Ireland is less severe than that reported in other countries (Gowen, 1990). The impact of widespread enrichment may however, cause the problems detailed by Frid and Mercer (1989) above.

A further cause for concern is the duration of these ecological and physio-chemical affects. Gowen *et al.* (1988) looked at the effect of a fish farm on the benthic community of a Scottish sea loch After the site had been used continuously for three years the sediment took eight months to revert to a state considered typical for the loch. Ritz *et al.*, (1989) investigated site recovery and demonstrated partial benthic recovery, in terms of species diversity, just seven weeks post-harvesting. Additional surveys at some Irish sites have shown that the sediment and ecosystem can recover from the effects of enrichment caused by organic waste from fish farms and in this respect confirm studies undertaken in other countries (Gowen, 1990). Johannessen *et al.* (1994) looked at the macrobenthos; before, during and after a fish farm. They discovered that the species number close to the farm declined from 65 in 1988 to only 11 in 1989. A year after the fish farm was moved, the number of species had risen to 29 thus indicating partial, but not complete, site recovery. Site recovery is also related to the disappearence of organic matter. Despite high organic and nutrient loadings, only 10% of organic matter in the sediment underneath salmonid farms is broken down annually (Aure and Stigebrandt, 1990). Waste accumulation, sediment and benthic recovery are all related to site conditions.

In order to protect the environment, monitoring must be carried out regularly to assess the effect of a specific farm. Ideally monitoring should be designed in a manner to trigger various mitigating measures at certain defined thresholds. In addition, monitoring results can be used to obtain information on longer-term effects (Dixon, 1986; Gowen, 1990).

1.3.3 Nutrient balances

Fish farms increase the input of nutrients into the system via feed and faeces. This can alter the nutrient balance within the system possibly resulting in a localised, or even widespread, change in ecology. Different proportions of nutrients will favour different primary producers. Gowen and Ezzi (1992) showed that fish farming contributed nitrogen to a Scottish sea loch thus altering the nitrogen to silicon ratio which is critical for phytoplankton production (see below).

Carbon

Findlay and Watling (1995) measured the benthic carbon flux at the pen edge of a salmon farm in Maine. The average flux was 5.0g C m^{-2} d^{-1}, a 1 - 6 fold increase over the ambient flux. Gowen (1990) estimated it as eight to ten tonnes of carbon released to the environment for every 50 tonnes of salmon produced. It is difficult to discuss these figures as the conditions and way in which they were calculated are not comparable.

Nitrogen

Gowen (1990) investigated changes in nitrogen levels at Irish fish farms and found minor changes in ammonium and nitrate levels at two individual farms but reckoned the scale of nutrient enrichment was unlikely to be ecologically significant. Apart from these cases he found no indication that fish farming in Irish coastal waters has caused any detectable change in levels of dissolved inorganic nutrients.

Hargrave *et al.* (1993) measured fluxes under salmon cages in the Bay of Fundy, Canada. They found that average rates of oxygen uptake and ammonium release for the three stations under the pens were 4 and 27 times higher, respectively, than values at the two control stations distant from the cages. Maximum average rates of ammonium release reached 38 mmol m^{-2} d^{-1} in late July.

Gowen and Ezzi (1992) noted localised nutrient enrichment in the immediate vicinity of the farm in the Sea Loch Hourn in Scotland. The highest concentrations coincided with neap tides when there was little flushing/water movement. Gowen (1990) calculated that during the production of 50 tonnes of salmon, three to five tonnes of soluble nitrogenous waste are lost to the environment. These losses are strikingly high if one considers that the loss of nitrogen alone may be 10% of the harvested weight of fish.

This excess nitrogen may affect the ecosystem. Of all nutrients, nitrogen would be expected to have the greatest impact, as in most coastal waters, including Irish waters, nitrogen is limiting. The release of soluble nitrogenous waste from fish farms could therefore stimulate phytoplankton growth (Gowen, 1990). Indeed, Gowen and Ezzi (1992) demonstrated that at the height of the growing season in a Scottish sea loch the ratio of nitrogen to silicate did change from 0.84:1 to 1.1:1. Given that phytoplankton require nitrogen and silicate in a ratio 1:1 they hypothesised that this could influence the species composition of phytoplankton and flagellates.

Phosphorous

It has been calculated that the phosphorous load from all Nordic fish farms is equivalent to the total Norwegian phosphorous load on the Skagerrak. This includes all point and non-point sources, such as municipal sewage water, countryside, agriculture, forestry, landfills etc. (Enell and Ackefors, 1991). Therefore, although one fish farm may have a small affect on the nutrient status of an area it is essential for management to take into account the cumulative impact of many farms. Environmental loads are presently calculated as 10 kg of phosphorous and 60 kg of nitrogen per tonne of fish produced (Ackefors and Enell, 1994). To date there is no evidence that fish farming has caused large scale hypernutrification and eutrophication of the tidally energetic coastal waters of Scotland and Ireland. According to Gowen (1990) eutrophication potential only exists in slowly flushed semi-enclosed embayments or straits.

Nutrients in the seawater near salmonid cages in the Bay of Fundy in Canada were investigated by Wildish *et al.* (1993). They noted that while levels of nitrate, phosphate, and silicate were not increased above the control, ammonia levels were higher near the cages. Hargrave *et al.* (1993) also reported ammonia, at a concentration of 62 mmol m^{-2} d^{-1}, measured under a salmon farm .

Food

Not alone do wastes, pollution and ecological effects originatefrom fish feed, but the harvesting of fish for feed also has ecological impacts. In many countries, in the past the principal feed used on fish farms was offal from slaughterhouses and fish offal. This was followed by reliance on daily landings of trash fish. Most farms have now switched to dry feeds (Jensen and Alsted, 1990).

In much of Europe one of the main sources of fish meal is the sand eel and other small sprat. This demand for sand-eels and other small fish (which are extensively used in the food pellets fed to caged finfish) may be having serious detrimental effects on sea-bird populations (O'Connor *et al.,* 1991). Arctic terns, kittiwakes, puffins, great skuas and red-throated divers have declined in population on the islands around Scotland and Shetland. Ornithologists believe that the cause of these deaths is starvation as a result of persistent industrial over-fishing of the sand-eel which is the staple diet of these seabirds. Salmonid farming, like other forms of intensive livestock rearing, relies on high protein feeds of which dried fish meal is a principal component (O'Connor *et al.,* 1991. In Britain, for example, the production of 28,000 tonnes of farmed salmon required approximately 35,000 tonnes of feed which was derived from a catch of 130,000 tonnes of wild fish (O'Connor *et al.,* 1991;

Scottish Wildlife and Countryside Link, 1990). It has been argued that salmonid farming contributes to overfishing of other fish species.

In Ireland, fish meal is produced primarily from the waste materials generated by fish processing plants; sand eels are not used. However, fish feeds incorporating wastes or offal have a much lower food conversion ratio (i.e. how many kg have to be fed in order for the fish to gain 1 kg) and their use leads to greater production of organic material and nutrients at fish cage sites. As a consequence they are being replaced where possible by dry low impact feeds. One such fish feed is produced in Ireland from 100% herring, compounded with high quality fish oil, vitamins, minerals and haemoglobin (O'Connor *et al.,* 1991).

Feeds also include phosphorous and vitamins, therapeutants and pigments. (Gowen and Bradbury, 1987). Colours are incorporated into salmon feeds to give the fish a flesh colour that is close to that of wild salmon. There are two colours; canthaxanthin and astaxanthin. Astaxanthin is the predominant colour used in farmed Atlantic salmon. It is synthetic and and is reported as possessing anti-cancer properties (Olsen *et al.,* 1989 cited in Bates, 1990).

Feeds are being further developed to have a low environmental impact. Ackefors and Enell (1994) noted that the amount of phosphorous and nitrogen in feeds has decreased to 1% and 7% respectively while feed conversion efficiency due to high energy feeds has improved to values around 1.2 in most salmon farming operations. Alternative food sources are being investigated (Hardy *et al.,* 1987) to break the dependence on other fish stocks. Feeding regimes are also being streamlined (Talbot and Hole, 1993). Profitability is often linked to feed consumption so waste minimisation is being profit-driven with positive knock on effects for the environment (Cho *et al*., 1994; Kolsater, 1995).

Feed dropping from the cages also has an impact on local fish populations. Carss (1990) noted that at marine farms the main fish species benefitting from the excess feed were saithe (*Pollachius virens*). This may serve to increase local fish populations and thus local predator populations which may have a knock-on effect on the farm. It addition it should be remembered that much of the feed is coloured and may be mediated with implications for the development of antibiotic resistance in local populations and thus disease control.

1.3.4 Escapees

Escapes may occur as a result of storms beyond the structural capabilities of the cages, inadequate management, predator damage to nets, vandalism and accidental damage (Brown *et al.,* 1990). Some hatcheries select for low thresholds for smolting and high thresholds for sexual maturation. Therefore major population characteristics are displaced toward the upper or lower extremes of genetic distributions (Thorpe, 1991). The intrusion of these fish into local populations could result in genetic changes which could compromise the fitness of the native population.

The potential scale of the problem may be quickly appreciated if one is to consider that in Scotland in 1987 the biomass of cultured salmon was 14 times that of native salmon and in Norway in 1989 estimate of cultured versus wild salmon was 60 to one (Maitland, 1989 and Hansen and Jonsson (1991) cited in Hutchings, 1991). In Ireland, in 1989 farmed production exceeded wild landings by over 6:1 (Browne, 1990). In Ireland the drift net five year (1991-1995) running average for salmon, was just 513 tonnes compared to 1,047 tonnes in 1986 (Wilkins, 1996). Farmed salmon production was 11, 811 tonnes in 1995 (M. Fitzsimmons, pers. comm.). If only 1% escape there is huge potential for dilution of native stocks (Browne, 1990).

This dilution has been observed. In Norway the proportion of reared salmon has increased from about 10% in 1986 to about 20% in commercial fisheries in Norwegian home waters in 1988 (Browne, 1990). In some localities even up to 30% reared fish have been reported (Mork, 1991). In 1987 brook stock surveys in a number of salmon rivers in Norway revealed that 23 out of 54 rivers (43%) examined contained reared salmon. A total of 615 salmon were examined and 83 (13%) were of reared origin (Gausen and Moen, 1991). Up to 78% of broodstocks examined in Norwegian rivers were of reared origin (Browne, 1990). Webb and Youngson (1992) looked at reared Atlantic salmon numbers in the catches of a salmon fishery on the western coast of Scotland. They found 22% were of reared origin and 65% were fish farm escapees. Overall in Ireland the occurrence of farmed fish in the catches is well below 1%. Unfortunately this is not the case in all areas. There are claims that 10%, and 20%, of salmon caught in Mayo and in Connemara respectively are derived from farmed stock (Browne, 1990).

Hansen *et al*. (1993) noted that escaped Atlantic salmon accounted for 25-48% in their samples at different stations in the oceanic waters north of the Faroe Islands. They also believed that in their samples were farmed fish of Scottish, Faroese and Irish origin. They expressed concern that this could lead to overestimates when assessing wild fishery stock. Not only have we a responsibility to conserve the genetic make-up of our own stocks but escapes in this country could ultimately affect those in another country hundreds of miles away.

There are two ways that escaped farm fish may impact the genetics of wild stocks: firstly they may compete for a limited spawning or rearing resource and displace, or partially displace, the natural populations and, secondly, the farmed fish could mate with wild stocks and produce hybrids (Peterson, 1993).

Hislop and Webb (1992) have shown that escaped salmon feed on natural prey in coastal waters. This means that they compete with the wild fish and are successful in their endeavours. Lura *et al.* (1993) demonstrated that spawning behaviour was intact in Noweigan fish farm escapees. However, competition for food and competition for spawning or rearing resources is likely to be won by the existing wild stock. This is because the existing stock has been naturally selected for the resources being contested (Peterson, 1993; Howell, 1994). Webb *et al.* (1991, 1993a and 1993b) documented an actual escape and return. They noted that in the same year, as the accidental release of 184,000 fish into Lake Eriboll, Scotland, angling catches in the Polla river, flowing into the lake, were 54% fish of farmed origin (Webb *et al*., 1991). The frequency of escaped farmed salmon in the River Polla, was estimated at spawning the second year after escape. There was no evidence of substantial returns of salmon from the large documented escape to these rivers. The findings suggest that more than 95% of those escaped fish which returned to rivers, near the site of the documented loss, did so in the first year after escape and that <0.5% of those fish which escaped returned within the two years of study (Webb *et al*., 1993a). They suggest that these farmed fish may have been subject to a high mortality at sea after their release. In Ireland, on the Burrishoole River, Co. Mayo, it has been shown that over a long period the return of reared fish is well below the return rate of their wild counterparts (Browne, 1990b).

The survival reared young fish escapees is low, but some may survive and migrate to spawning grounds with wild stocks. The adult escapees may spawn in rivers adjacent to the farm site, but timing is critical as these fish are subject to predation and other environmental hazards and are unlikely to survive in the wild for an extended period (Peterson, 1993). However, these suggestions and assumptions are poorly borne out by the high percentages of farmed fish in Norwegian waters. This may be because most cultured fish are of native Norweigan origin and thus these fish had the necessary qualities to help them survive. This may not be the case for escapees from Irish farms which would mainly be of Norweigan and Scottish origin (Browne, 1990).

Survival of reared salmon in the wild will depend on size and age of the salmon, the site of escape, the time of escape, physiological status and domestication (Hansen, 1990). Fish that are released or escape from a fish farm at smolt stage in freshwater will return with high precision to that site to spawn (Hansen, 1990; Egidius *et al*., 1991). Adults tend to stay close to the farm site while younger fish tend to disperse (Peterson, 1993). The chance of straying increases if they escape from an estuary and when smolts escape at a marine locality they return to this area when mature and will enter rivers in that area to spawn (Hansen, 1990).

Adults escaping in summer seem to behave like homeless fish, and enter the rivers at random for spawning (Egidius *et al.*, 1991). Irish experience would suggest that fish escaping at or near spawning time will tend to enter local rivers to spawn. Long-term survival depends on size, the maturity of the fish and the time of escape (Browne, 1990; Saunders, 1991). Farmed fish appear to enter rivers later and spawn later than their wild counterparts. This suggests that they are indeed different to their wild counterparts. They have been known to overcut redds and physically displace wild eggs (Browne, 1990; Saunders, 1991).

The frequency of escaped farmed salmon spawning in the River Polla, Scotland was estimated in 1990, the second year after the escape (Webb *et al*., 1993a). Fourteen of the 73 spawners examined were of farmed origin. Only six of these fish were identified as being part of the documented escape. All of these carried the pigment canthaxanthin (Webb *et al*., 1993a) which is added to some farmed foods to colour the fish flesh pink. Five of fifty five redds, which were constructed for spawning, were sampled and found to contain alvins or embryos bearing canthaxanthin (Webb *et al*., 1993a). This sucess rate is significant and it should be remembered that in Norway several spawning populations of fish of cultured origin now exceed the number of wild fish (Heggberget *et al*., 1993). However, Oklan*d et al.* (1995) suggest that escaped farmed salmon had reduced spawning success compared to wild fish.

Webb *et al*. (1993b) investigated the egg production from escaped farmed Atlantic salmon, *Salmo salar*, in western and northern Scottish rivers. They detected canthaxanthin in 14 of 16 rivers sampled and in 5.1% of fry sampled. As males do not contribute to the canthaxanthin marker a figure of 7% is suggested as more realistic although it is still an underestimate as freshwater salmon escapees would have been excluded. This is because they would not have been fed this pigment until they were transferred into the salt water phase. From their results survival amongst the farmed salmon seems low. The figure of 7% of the fry in the river would be best set against the survival of this generation. Even when spawning and fertilisation is successful, Peterson (1993) states that laboratory work has shown that offspring from these crosses have low viability and are sterile. This means that they could possibly displace local populations or establish feral populations in a vacant environmental niche, but they will not contribute genetically to native stocks. Skaala (1994) demonstrated that the farmed fish had a lower reproductive success than the released fish and that their relative reproductive success was 25-30% of the wild fish. In contrast, Lura and Saegrov (1991) confirmed that in their study, also in Norway, that the success of farmed females is similar or slightly lower, as compared to wild spawners in the same area. They concluded that gene-flow cannot be neglected as an influence on wild stocks.

Interbreeding between escaped farmed salmon and wild Atlantic salmon (*Salmo salar*) in a Northern Irish river has been observed by Crozier (1993). He noted that allele frequency comparisons indicated that following the escape of farmed salmon the genetic composition had become more like that of the presumed escapees, although remaining statistically significantly different (Crozier, 1993). Clearly the wild population had been demonstratably altered by the escapees spawning in the river. The presumed escapees were 93% ripe or ripening males and it was likely that they interbred with wild females (Crozier, 1993). This type of intrusion could have disaterous effects on native locally adapted populations of fish.

The threat of extinction to native populations subject to spawning intrusions by cultured Atlantic salmon was investigated by Hutchings (1991) using a model to predict the outcome of various scenarios. The model demonstrates how spawning intrusions by cultured salmon can result in the rapid reduction and eventual extinction of native genomes. The magnitude and degree of permanence of these reductions depend on the frequency of the spawning intrusions and on the proportion of the spawning population comprised of cultured individuals (Hutchings, 1991). Small, native populations face the greatest threat of extinction through the swamping of their genome and of their habitat by cultured salmon (Peterson, 1993; Hutchings, 1991). Hutchings (1991) cautions that the model may underestimate the ability of the cultured population to adapt to local conditions. It is not clear that cultured salmon would possess sufficient additive variability for fitness-related traits to enable them to respond rapidly to natural selection (Hutchings 1991). In contrast, some biologists maintain that restocking for enhancement and restoration purposes over many years has had no demonstrable effect on the fitness of wild salmon (Saunders 1991). Some geneticists suggest that such interbreeding could actually benefit wild stocks as it may contribute additional valuable genetic information (e.g. see Peterson, 1993; Thorpe, 1991). Hutchings (1991) urges the error to be on the side of conservation.

1.3.5 Disease

Wild fish carrying pathogens but not exhibiting the disease are a major cause of disease outbreaks on fish farms (McArdle, 1990). While it is well established that wild fish pose a threat to farmed fish there are very few cases of the reverse (McArdle, 1990; Saunders, 1991).

Bacterial

Furunculosis is endemic in the wild populations of Scotland, Ireland and Canada, but is rare in free living fish. In fish farms the disease can be disastrous. It seems that wild fish are the source of infection and outbreaks on farms seem to be associated with fish density in tanks, cages and other stress inducing factors (Mitchell, 1992). Its causative agent is *Aeromonas salmonicida* (Saunders, 1991).

Bacterial kidney disease (*Renibacterium salmoninarum*) similarly is troublesome on fish farms (Saunders, 1991). Although wild salmon are suspected as the source, little is known to what extent wild salmon are at risk through contact with infected cultured salmon (Saunders, 1991). There is no evidence of transmission from farmed to wild fish (Brackett, 1991). In contrast, for Infectious Pancreatic Necrosis (IPN) there has been documented transmission from wild to farmed and vice versa (Brackett, 1991). While examples of disease interactions between wild and farmed fish can be found under many different conditions, documented disease interactions are not as common as some might fear (Brackett, 1991). The freshwater fluke *Gyrodactylus salaris* has done great harm in Norwegian rivers, but in the fish farming industry it is not a problem. This is the only disease where wild stock are the losers (Egidius *et al*., 1991). However as the fluke dies in seawater it is beyond the scope of this report.

Parasites

Sea lice, *Caligus elongatus* and *Lepeophtheirus salmonis,* do not survive in freshwater. Although wild fish are sometimes heavily infested with sea lice these parasites rarely cause death or serious injury in nature. Cultured salmon are most likely infested with sea lice. These mobile parasites move from wild fish near the sea cages. According to Saunders (1991) sea lice pose a greater threat to cultured as opposed to wild salmon and confinement encourages heavy infestations.

Cultured salmon are frequently more susceptible to disease than their wild counterparts. This is a result of confined, often suboptimal, conditions (in terms of water flow, temperature and dissolved oxygen) (Saunders, 1991). Under intensive conditions found in farms *Lepeophtheirus salmonis* larvae are in close proximity to their specific host and so it is relatively easy from them to complete their lifecycle and find new hosts (Egidius *et al.*, 1991).They live on mucus, skin and blood and in the most advanced cases fish are seen with large open lesions, especially in the head region. Wild salmonoids have been observed carrying large numbers of lice but lesions or mortality in such fish have not been reported (Egidius *et al.*, 1991).

In 1992 and 1993 a statistically significant relationship was shown between lice (particularly *Lepeophtheirus salmonis*) infestation on sea-trout and the distance to the nearest salmon farm (STWG, 1992, 1993). Sea trout populations are falling and sea lice infestation from salmon farms is being blamed in some circles. The Burrishole River, Co Mayo has been studied intensively with 6,710 smolts migrating downstream in 1981 and 1,698 in 1993. The upstream run of finnock and adult sea trout was 3,200 in 1976 and slumped to around 155 from 1990 to 1993. The marine survival of trout smolts returning as finnock in the same year, which historically ranged from 11.4% to 32.4% fell to 1.5% in 1989. Despite some recovery in 1990/1 the survival rate for 1993 was just 6.2%. Tinley (1993) noted that the sea trout catches had dropped from 9,000 fish in the 1950's to around 2,000 in recent years and described this trend as worrying. The Seatrout Working Group (STWG) (1992) investigated physical, biological and chemical factors to determine the cause of the seatrout collapse. The most up-to-date findings of the group are embodied in their annual reports between 1992 and 1994. McVicar *et al.* (1993) commented that this trend was also seen in western Scotland. In 1994 no significant relationships were observed. The relationship was seen to have become positively weaker over the three years, coincidental with reduced lice levels on farms. The results to date do not yet constitute conclusive proof of a causal link between sea lice infestations of farmed salmon and infestations on wild sea trout in nearby rivers (STWG, 1992, 93 and 94)

Bass and Murphy (1995) argue that sudden whole shoal infections are not characteristic of fish farm infestation by *Lepeophtheirus salmonis* but more typical of *Caligus elongatus*. In contrast, Costelloe *et al*. (1995) concluded from their studies on the variation of lice infestation on salmon farms that the source of lice can be from wild or farmed stock but that significantly higher concentrations of lice infestation in the inner estuary in Killary Harbour indicate that lice larvae originated from wild fish. Good management practices coupled with careful chemical treatments can help to control lice on fish farms.

1.3.6 Chemicals

The main concerns associated with releasing chemicals into a marine system were addressed by Redshaw (1995). According to Redshaw and other authors (see below), possible effects include direct toxicity to non-target organisms, uptake of contaminants by wild fish and shellfish, inhibition of microbiological activity in the sediments below fish cages (thereby affecting the rate of degradation of accumulating organic matter), induction of antibiotic resistance in aquatic organisms including fish pathogens, and with concomitant effects on humans (Smith *et al.*, 1994).

Orthophosphates

Dichlorvos, or Nuvan (or Aquaguard) as it is better known, is an organophosphorous pesticide, widely used in the agriculture industry. It works by inhibiting nerve transmissions (Anon., 1989). Sea lice are normally controlled by the use of Dichlorvos or sometimes by Ivermectin (Kennedy *et al.,* 1993). According to Ross (1989), given dichlorvos' known toxicity and the extensive gaps in our knowledge about its environmental effects, the widespread and growing use of this substance in the marine environment is unacceptable. Furthermore, the current controls are inadequate and contrary to the precautionary principle enshrined in EU law (Ross, 1989).

The greatest concern is that this chemical is toxic to arthropods and has potential effects on marine invertebrates. Little is known about its rate of breakdown in the marine environment (Ross , 1989). Cusack and Johnson (1990) looked at the effects of Dichlorvos on various marine species. Their results indicate that Dichlorvos is not toxic to mussels or periwinkles at 1 ppm for 1 hr exposure but is toxic to larval lobsters, adult lobsters, zooplankton and phytoplankton. However, in field trials larval and juvenile lobsters were not killed following treatment with Dichlorvos. This is probably due to the more realistic treatment levels, coupled with the rapid dilution. Thain *et al.* (1990) found that organisms tested reacted negatively to some level of Dichlorvos. The marine algae were relatively insensitive and of the bivalves tested (*Crassostrea gigas*, *Venerupis decussata* and *Mytilus edulis*) *Crassostrea gigas* was the most sensitive. The LC 50 (lethal concentration killing 50%) was 4.4, 11. 1 and 14.4 μgL^{-1} for Brown Shrimp (*Crangon crangon*), *Msidopsis bahia* and Common limpet

(*Patella vulgata*) respectively. Le Bris *et al*., 1995 noted that while Dichlorvos caused no abnormal mortality in Manila clam or Pacific oysters, it caused them to open due to relaxation of the adducter muscle. This may result in increased successful predation. Dobson and Jack (1991) evaluated the dispersion of treatment solutions of Dichlorvos from marine salmon pens. They discovered that there was no detectable level of Dichlorvos found outside a 25 m perimeter of the treated pens at any time. Twenty four hours later there were no detectable levels of Dichlorvos either within the pens or beyond 25m. On one hand this literature shows that Dichlorvos can be lethal to many organisms and cause changes in behaviour in others, however the quantities involved are greatly diluted in the marine environment.

Ivermectin is an alternative to Dichlorvos, it has been legally used in Ireland for years in order to control sea lice, it has the advantage of being less toxic than Dichlorvos but has the disadvantage of being highly persistant. It works by paralysing the neuromuscular system of invertebrates and there is concern that it will accumulate in the food chain reaching the consumer (Edwards, 1996). In addition, shellfish in the vicinity of the farm are likley to accumulate large concentrations. Laboratory tests have demonstrated that Ivermectin is lethal to crustaceans and to the lugworm *Arenicola marina* (Edwards, 1996) however, the Ivermectin concentrations tested were not reported.

The single bay management plan makes provision for co-ordination of treatments (STWG, 1991, 1992 and 1993), so that infestations such as sea lice are wiped out together and there is no source for repeat infestations. However, this will result in a greater quantity of chemicals than normal reaching the environment with potentially harmful ecological effects. This may have a beneficial affect on the sea trout but is likely to have a negative effect on some other marine organisms.

An alternative solution to using Dichlorvos for treating salmon infected with lice is to use wrasse. Wrasse are "cleaner-fish" that feed on sea lice present on the salmon. Deady *et al*. (1995) demonstrated that wrasse and goldsinny were a more effective lice control than conventional chemical treatments. However, they warn that the possibility of overfishing wrasse in smaller bays considerable. Tully *et al.* (1996) showed in their experiment that wrasse (*Ctenolabrus rupestris* and *C. exoletus*) failed to control infestations of the sea lice *Caligus elongatus*. However the use of this method of control has expanded greatly in many years, particularly in Norway, thus indicating the general success and practical application of this control method.

Antibacterials

Bacterial diseases such as furunculosis are treatable with antibiotics such as oxolinic acid and and oxytetracycline. However, there are some drawbacks associated with their use. The main concerns are antibiotic resistance and retarded degradation of wastes.

Antibiotic resistant bacteria have been recorded in hatcheries (Olivier, 1992; Smith *et al*., 1994), in fish feed and in sediments beneath farms (Kerry *et al*., 1995; Duis *et al*., 1995). The problem with resistance is that the target organisms no longer respond to the antibiotic treatment making it increasingly hard to fight infection. Changes in husbandry are necessary to limit the rate at which resistant strains emerge, otherwise we may re-enter the pre-antibiotic era (Smith *et al*., 1994). As only oxytetracycline and oxolinic acid were licensed for use in Scottish fish farms, massive use of the latter for control has resulted in widespread resistance (de Kinkelin, 1992). In addition, Callanan (1996) and Smith *et al.* (1994) observed that microrganism susceptibility tests are often prone to error and the application of these results can have serious ecological consequences as ineffective antibiotics are being added to the marine environment killing non-target, beneficial microorganisms (Johnson, 1994). Some evidence suggests that antibiotics persist in sediments (Jacobsen and Berglind, 1988; Bjorkland *et al.,* 1990; Bjorkland *et al.,* 1991; Pouliquen *et al*., 1992; Coyne *et al*., 1994). Misuse of antibiotics has implications not only for fish health but for human health also. De Kinkelin (1992) noted that Norway used 50 tonnes of antibiotics in fish farms in 1990, about double that used in human medicine for the same year.

Antifoulants

Antifoulants are used in salmonid culture to prevent growth on the nets (Minchin *et al*., 1987; Lau, 1991) and so allow a good through flow of water. Until recently Tributyltin (TBT) was used and this has caused problems as it is a bioaccumulator (Lau, 1991), causes imposex in dogwhelks and and abnormal shell growth in oysters (Minchin *et al*., 1995). Lau (1991) investigated the Hong Kong marine environment and found that more than 90 of the sampling sites registered levels exceeding 90 ngL^{-1}.

In fact, the use of TBT on salmon cages in Mulroy Bay, Co. Donegal is probably responsible for one of the most severe environmental impacts attributable to aquaculture in Ireland. In Mulroy Bay (Co. Donegal, Ireland) scallop spat settlement had occurred every year since 1967 (Minchin, 1981b). There were poor settlements or settlement failures over the period 1981-1985, possibly due to the use of organotins at the time (Minchin and Ni Donnachada, 1995). The use of organotins as fouling agents on cage nets coincided with this period and moderate settlements returned once the use of organotins ceased (Minchin, 1995). The flame shell *Lima hians* in Muroy bay also suffered, with no recriutment for two years, as a result of TBT use on salmon cages. The population declined and the interwoven byssal threads from this animal, which stablised the sediment, was lost leading to sediment instability. Currently the antifoulants used on salmon cages are copper based (C. Duggan, pers. comm.). There is no evidence in the literature of the toxicity of these compounds.

1.3.7 Predators

Birds

According to the Scottish Salmon Growers Association (1990) the avian predators on fish farms are grey herons, gulls, terns, cormorants, shags and other species of diving birds. A 1985 survey of Scottish finfish farms revealed that many farmers considered piscivorous birds a problem and that killing of such birds was widespread practice (Carss, 1994). Herons, cormorants and shags were the most commonly reported predators at 72% of farms surveyed. Carss (1994) estimated that 800 herons, 1,643 cormorants and 1,405 shags could have been killed each year at Scottish finfish farms (1984-87).

Herons typically prey on salmon farms by landing on the top net and reaching into the cage through the mesh, occasionally attacking from the walkway. The majority of damage occurs at dawn when the birds are most active, increasing again in the evening (SSGA, 1990; Carss, 1993). Carss (1994) looked also at the method of killing. Adult herons mainly died by drowning in cages after falling through holes in the top netting which were sometimes made deliberately for this purpose. Proportionately more adults were killed at cages (Carss, 1994). Previous studies have shown that killing does not reduce the numbers of herons visiting farms (Carss, 1994). However, as 800 herons represent only 7% of the Scottish mid-winter population this persecution may actually have little effect on the overall national population (Carss, 1994).

Cormorants dive close to the side of the cages and spear the fish through the net. It is unlikely they can extract fish this way but they can cause considerable damage (SSGA, 1990). Carss (1994) noted that most cormorants killed at fish farms were shot. Some cormorants and shags occasionally drowned in underwater nets but numbers were small (4% and 7% respectively) compared to those killed deliberately. The estimated 1,643 cormorants killed annually represent 18-23% of the Scottish population and as the majority killed were mature this represents a real and significant loss to the population. Carss (1994) noted that decreases in breeding populations of cormorants corresponded to those areas with the highest density of fish farms where as where there were few farms, numbers are actually increasing. All the shags (1,405) killed were immature and so Carss (1994) concludes that it is unlikely that persecution would have any long-term effect on the Scottish population. Murphy (1992) surveyed 12 farms in Ireland and discovered that a total of 129 herons, 16 cormorants and shags and 10 gannets were shot since the farms were established.

Other species such as gulls, terns and auks are reported as frequenting fish farms. Five species of gull may frequent fish farms; blackheaded, common, herring and great blackbacked gulls throughout the year and lesser black-backed gulls during the spring and summer. They generally feed on waste food, or unprotected food on fish farms (SSGA, 1990). The arctic and common tern are both reported as visitors to fish farms. At the farms they take small fish (less than 10 cm) which they take by plunge diving (SSGA, 1990). Gannets rarely have been reported as behaving similarly (SSGA, 1990).

Auks such as puffins, black guillemots, razorbills, divers (red-throated, blackthroated and great northern) and red breasted mergansers are other predators which may visit fish farms. These birds do not usually feed on caged stock but may be attracted by an increase in wild fish in the vicinity of the cages and so may become entangled in anti-predator nets (SSGA, 1990).

Various netting strategies are available to exclude predators. The flat top net tensioned over the cage protects against herons and other surface feeders. Walk-under top nets also provide protection to food supplies. Underwater nets can be effective against diving birds and seals (SSGA, 1990). The Scottish Salmon Growers Association (1990) states that setting nets with the objective of entangling predators is illegal. This indicates that these practices have happened in the past and may continue to happen.

Other alternative methods of control are acoustic bird scarers. However, it is generally accepted that habituation is common and they provide only temporary relief (SSGA, 1990; Behrendt, 1995). One sucess non-lethal method was discovered by one farm which sucessfully used falcons to scare herons away (Behrendt, 1994).

Seals

Scottish fish farmers are estimated to lose £3 million to seal predation each year (Anon., 1995). Seals attack nets squashing the fish between flippers and sucking viscera through the net. Not alone are the fish, eaten or wounded, lost but there are other losses to the farmer as stressed fish are more susceptible to disease and often lose their appetite. The number of seal attacks appears to peak in winter post breeding and post moulting (C. Crummy, pers. comm.). Losses from seal attacks can be in tonnes as the nets often become torn. Pemberton and Shaughnessy (1993) looked at the interaction between seals and marine fish-farms (*Salmo salar* and *S. gairdneri*) in Tasmania, and management of the problem. They noted that seals usually attacked cages at night. The seals damaged both the pens and the fish and sometimes fish escaped as a result of the attacks. The vulnerability of the fish farms was influenced by their proximity to haul out sites.

The farmers tend to tackle seal predation in a number of different ways. Acoustic seal scarers are one method of protection. The success of this method is very variable (Murphy, 1992; Pemberton and Shaughnessy, 1993; Anon., 1995) and in order to be be effective at least six would be needed for the average sized farm making their use very costly (Pemberton and Shaughnessy, 1993). Pemberton and Shaughnessy (1993) noted that even so, the level of attack on farms with acoustic scarers was unacceptable. They recorded many attacks in the presence of acoustic seal scarers. However, more powerful systems have been developed (Anon., 1994b). The scarers have been shown to be effective against both seals and birds, but only for a short length of time due to habituation (Ministry of Agriculture, 1991). Some concern has been expressed about the possible effects of acoustical seal scarers on seals and other wildlife (SSGA, 1990).

Another repellent method, which is certainly environmentally friendly, is the deployment of a 17 ft Killer Whale model close to the fish farms. Natural instinct keeps the hungry seals at bay and this project proved very successful in Canada (Anon., 1994a). However, the same success was not achieved in Ireland. Exclusion methods such as tensioned predator nets have also been tried. However, these have also proved unsuccessful (C. Crummy, pers. comm.).

Unfortunately some fish farmers are recorded as opting for lethal methods of control. This is not documented for Ireland. In Scotland the deliberate disturbance and the destruction of local seal colonies by boat has been noted (SSGA, 1990). However, this is considered to be totally unacceptable by the industry itself (SSGA, 1990). Licence to shoot 200 seals has been given to the salmon farmers, in Canada, since 1994 (Wilbur, 1994). Pemberton and Shaughnessy (1993) noted that the use of shooting as a protection method was inefficient and ineffective, because seals usually enter the fish farms at night and show no fear of shooters. They recorded many attacks continuing during shooting. In Ireland Murphy (1992) surveyed 12 farms and discovered that a total of 364 seals were shot since the farms were established.

Pemberton and Shaughnessy (1993) also noted that deterrents such as pursuit with boats, lights, underwater explosive crackers and emetics to induce conditioned food aversion helped reduce the number of seal attacks. They concluded that the only way to prevent seals from attacking fish farms is to exclude them from the vicinity of the fish pens with physical barriers they cannot penetrate. These are currently in use, and include perimeter fences and protection nets made of steel mesh set around individual pens. They recommend that seal shooting permits only be issued to farms that have taken adequate precautions to prevent seals attacking fish in holding pens. Regarding seals, no totally effective method of exclusion has been identified (SSGA, 1990). This conflict is likley to increase as the UK seal population is growing at a rate of 7% per annum (this is probably also true for the Irish seal population)and is estimated at 80,000 individuals (Crummy, 1993).

It must be remembered that the factors which control populations size of wild animals are predator numbers, habitat availability and food supply, therefore lethal methods of control are bound to prove ineffective as these determinants are unaffected.

Other predators

The Scottish salmon growers association (1990) and Murphy (1992) mention predation by otters but the levels of predation by otters seems low. Two otters were reported shot by Murphy (1992) in the 12 farms he surveyed. It must be borne in mind that although otter populations are in a healthy state in Ireland they are an internationally important species, extinct throughout a lot of Europe, and as such must be afforded the greatest protection.

1.4 Summary

Nutrients, Waste and Fish Feed

Organic waste derived from excess food and faecal materials accumulate under fish farm cages. This waste is oxidised resulting in oxygen depletion and negative redox potential in the sediment. Anaerobic bacteria ferment the waste producing hydrogen sulphide and methane. High numbers of opportunistic polychaetes colonise the enriched sediment under and/or adjacent to the farm. In some cases the change in benthic fauna can be detected up to 150 + metres from the farm boundary. Every site is different and environmental quality standards are proving difficult to set. Sheltered sites may result in higher waste accumulation, but well flushed sites have the potential to lead to phytoplankton blooms. Nutrient balances may be affected by finfish farming with concomitant effects on the ecology of the surrounding marine environment. The feeding of wild fish to farmed fish in the form of fish feed may have knock-on effects on the interdependent species.

Escapees

There is a huge potential for swamping native genomes with farmed ones and caution must be exercised.Escaped fish have been shown to eat natural food sources, find their river of origin, and successfully reproduce with wild salmon. Although it is unknown how well escapees survive.

Disease and Control

Wild fish can transfer disease to cultured ones but the reverse transfer is still under investigation.In both 1992 and 1993 a statistically significant relationship was shown between lice, infestation on sea trout and the distance to the nearest salmon farm. The relationship was not shown to hold in 1994..

There are potential problems with non-target effects of the chemicals Dichlorvos and Ivermectin which are used to treat sea lice infections on salmonids. An alternative for lice control is wrasse. Results are conflicting, but given the success of this method in Norway there are possibilities that it could be used here. Antibiotic use is tainted by increasing resistance leading to difficulties in controling disease.

Predators

Herons and cormorants are the main avian predators on fish farms. Shooting is ineffective and tensioned exclusion nets appear the only option. Seals are serious predators on finfish farms and methods of control are discussed.

1.5 Management

There are two strands to management of finfish for ecological protection. The first is to select sites which are suitable and the second is to manage the operation in such a way as to cause minimum ecological disturbance.

Prevention is better than cure and site selection is the most important, efficient and effective way of ensuring that the ecological impact is minimal. Sites should be selected in terms of water exchange (Gowen *et al*., 1988; Gowen, 1990), predator numbers (SSGA, 1990), sediment type (Lumb, 1989) and distances away from salmon rivers (Browne *et al*., 1990; STWG, 1991, 1992 and 1993). For all considerations on site selection see Guidelines for selecting a fish farming site (Anon., 1991a; Levings, 1994) and strategies for regulation of aquaculture site selection in coastal areas (Black and Truscott, 1994).

Once a farm is operational certain conflicts may be inherent in the siting. Therefore the onus is on the farmer to manage the farm in as environmentally responsible a manner as possible(Anon., 1991a). It must be stressed that this type of day to day farm management requires commitment and initiative from the farmer. A grass roots approach to environmental management is necessary in order to secure a sustainable industry. Whereas management prescriptions are valuable, they are only as good as the people who employ them. There must be incentives for the farmer to operate in an environmentally aware manner, be it via recognised environmental systems such the ISO 14,000/IS310 or an eco label.

1.6 Mitigating measures

To minimise waste accumulation there should be:

- Optimisation of feeding strategies (Rosenthal, 1994; Alanara *et al.*, 1994; Makinen, 1991).
- Improvements in feed formulations (see example Talbot and Hole, 1994).
- Development of efficient mechanical methods to remove wastes e.g. funnel/cone shaped collectors under the cages (see example Behmer, 1993; Ackefors and Enell, 1994).

To remedy waste accumulation there should be:

- Harrowing under a salmon farm as this has been shown to be highly effective (O'Connor *et al.*, 1993).
- Fallowing to aid recovery of the seabed (Dixon, 1986; Ministerial Advisory Group on Fallowing, 1993).

To minimise use of chemicals there should be:

• Improved husbandry and health management of farmed fish, for example, separation of different salmon classes (Rosenthal, 1994; Anon., 1992).
• Restricted movement of fish (Browne *et al*., 1990).

To minimise the chance of fish escaping there should be:
• More effective containment (Hindar *et al*., 1991).

To minimise impact of escapees there should be:
• Exclusive use of local stocks in breeding programs (Hindar *et al*., 1991, Mork, 1991).
• The utilisation of monosex or sterile populations of fish may be a solution or partial solution for the problems associated with sexual differences, sexual maturation, and unwanted reproduction (Skaala, 1994). Although more effective containment is stillpreferable to sterilisation for ecological reasons (Hindar *et al*., 1991).

To minimise impact on predators there should be:
• Good management practices such as careful waste management and food storage.
• Every reasonable measure should be taken to exclude predators (SSGA, 1990).

Best environmental practices for controlling predators:
1. Various netting strategies are available to exclude birds (SSGA, 1990; Anon., 1991d; Pemberton and Shaughnessy, 1993).
2. Acoustic seal/bird scarers are one method of protection. The success rate however, is very variable (Anon., 1991d; SSGA, 1990; Pemberton and Shaughnessy, 1993).
3. Pemberton and Shaughnessy (1993) noted that deterrents such as pursuit with boats, lights, underwater explosive crackers and emetics to induce conditioned food aversion helped reduce the number of seal attacks.
4. Pemberton and Shaughnessy (1993) concluded that the only way to prevent seals from attacking fish farms is to exclude them from the vicinity of the fish pens with physical barriers they cannot penetrate.

Not all of these measures will be appropriate to every site. The methods used must be practical, as user friendly as possible and highly effective in order to ensure a high take up.

OTHER FINFISH CULTURED IN IRELAND

Seatrout (*Oncorhynchus mykiss*)

In 1995 470 tonnes of seatrout were farmed in Ireland. Seatrout are much bigger than freshwater trout. They are usually grown to 1 kg and compete in the market with salmon (Anon., 1993). The manner in which they are grown is similar to salmon. From the literature it would be expected that the impacts are very similar. There are very few references to the environmental impacts of their production.

The only documented impacts relate to nutrient fluxes. Hall *et al. (*1992) discovered that of total nitrogen input to the farm 27-28% was recovered in harvest and the nitrogen lost was 95 to 102 kg N per tonnes of fish produced. The order of magnitude was the same for salmon production as calculated by Gowen (1990) and Ackefors and Enell (1994). Holby and Hall (1991) calculated that for each tonne of trout produced, in a marine fish farm in Sweden, between 19.6 and 22.4 kg of phosphorous was lost to the environment. Again this was the same order of magnitude as calculated for a salmon farm by Ackefors and Enell (1994). Hall *et al.* (1990) measured the benthic carbon flux originating from a marine cage farm in a typical fjord in western Sweden. They found sedimentation rates of 29.7-74.9 g C m^{-2} d^{-1}, greater than the control during peak periods of trout production. This contrasted greatly with Findlay and Watling's (1995) calculation of 5 g C m^{-2} d^{-1} for a salmon farm. However, as withall these calculations this could be a site specific result. In 1994, Hall and Holby discovered that the loss of silicon to the environment was between 2.4 and 2.5 kg (values differed for different years) for every ton of trout produced. There were no comparable figures for silicon release on salmon farms. Holmer and Kristensen (1992) reported that sediment metabolism under a marine trout farm was 10 times higher than the control. Increased sediment metabolism was also reported under salmon farms (Hargrave *et al*., 1993). Given the comparability the results and of the species and their culture conditions, it would be surprising if the impacts were very different.

Halibut (*Hippoglossus hippoglossus*)

Trials with halibut have only commenced in recent years, and are ongoing at Cape Clear Island, Co. Cork. The current growing system is an onshore systems facility with sea water pumped into it. To date the trials have been successful and there is some consideration being given to using modified salmon cages for production (Anon., 1995). No references specific to halibut production were discovered.

Turbot (*Scophthalmus maximus*)

Turbot is being farmed successfully also at Cape Clear Island, Co. Cork. In 1995/5 16-17 tonnes of turbot was harvested this is likely to be in the region of 25 tonnes for 1996/7. They intend to increase their production capacity to 50 tonnes by 1997. As for halibut, the system is an onshore facility with sea water pumped into it (Anon., 1995). Turbot showed a low metabolic activity confirmed by low nitrogenous excretion levels (Dosdat, 1995). There are three other experimental turbot farms: two in Cork and one at Tarbert in Co. Kerry.

Management

The mangement strategy for trout will be similar to that described for salmon given the current state of knowledge. However turbot and halibut are farmed in a completely different manner and research will have to identify the environmental impacts associated with them as well as the optimim modes of management for these species.

SECTION 2

SHELLFISH CULTIVATION I

MUSSEL (*MYTILUS EDULIS*)

1.1 Mussel cultivation in Ireland

The only mussel cultured in Ireland is the native mussel, *Mytilus edulis*. It is widely distributed and adapts itself to a wide variety of ecological situations. Mussels can tolerate fluctuating salinities but grow best between 20-30 ppt. Mussels are filter feeders, feeding on phytoplankton and suspended organic matter, thus feeding entirely on natural food present in the water column (BIM, 1982). This reliance on a natural food source, coupled with their general sessile nature, makes them ideal for cultivation. In Ireland, two distinct methods of mussel cultivation are practised; suspended and bottom culture. These will be looked at separately.

1.2 Data available

Mussels (*Mytilus edulis*) both in extensive and intensive culture are perhaps the most intensively studied species of all the cultivated shellfish. Fortunately much of this research has been carried out in comparable northern temperate climates such as in the Dutch Wadden Sea (e.g. Dankers and Zuidema, 1995). Subsequently little further general research is needed on its environmental impacts. However, more research may be needed to address specific questions relating to particular sites.

1.3 Bottom cultivation

Bottom culture accounts for about half of all mussels produced in Ireland. In 1995 5,570 tonnes were produced. Bottom cultivation involves the location, collection and transplantation of wild mussel spat into richer, shallower waters using a dredger. Successful ongrowing of re-laid spat requires sandy shallow beds. When the mussels reach commercial size (9-18 months later), they are harvested by dredger (Joyce, 1992). This method is practised successfully on a large scale in Wexford Harbour and also in Carlingford Lough.

1.4 The impacts of bottom cultivation

The impacts of shellfish cultivation will depend not only on the species being cultured but the characteristics of the site and on farm management practices. The impacts can be looked at from two angles; localised impacts (immediate farm area) and regional impacts (bay or geographical area).

LOCALISED IMPACTS

There are many ways in which the area immediately surrounding the farm is affected by its presence. These impacts will be looked at under the following headings:
1.4.1 Water composition.
1.4.2 Biodeposition and nutrient recycling.
1.4.3 Biodeposition and the benthos.
1.4.4 Competition and space .

1.4.1 Water composition

Normally any changes in water composition can only be detected immediately beside the farm due to dilution by the sea. Changes in water composition are mainly due to removal of suspended solids from the water and excretion of soluble waste products back into it. In the western Wadden Sea mussel beds have been shown to take up total suspended sediments, chlorophyll *a*, total organic carbon, nitrite and nitrate (Dame and Dankers, 1988; Dame *et al*., 1991). Dame *et al.* (1991) discovered that the mussel bed sediments are also sources of nitrite and nitrate. Dame and Dankers (1988) noted that the mussel beds themselves released a significant amount of ammonium and *ortho*-phosphate. Dame *et al.* (1991) found that silicate was also released and that seston or total suspended particulate material was both taken up and released by the mussel beds. The general trend for silicate release from mussel beds is probably the result of phytoplankton cells breaking down as they are metabolised by the mussels (Dame *et al*., 1991).

This release of nutrients by the mussel beds is a recycling mechanism and may help to offset the nutrient loss to the system. Asmus and Asmus (1991) considered whether ammonium release by a mussel bed could act as a promoter of phytoplankton. They concluded that the released ammonium could be utilised by phytoplankton, if no other nutrient compound were to become limiting. In some natural beds, mussels and *Fucus* spp. co-occur. In this situation these algae benefit directly from the release of nutrients by the mussel bed. Thereby the different ecological role of natural mussel beds as compared to cultured mussel beds without algal cover becomes apparent. Mussel beds with algal cover have a strong tendency to keep nutrients within the benthic system, where as bare mussel beds release nutrients into the overlying water. Asmus and Asmus (1991) concluded that in culturing systems the release of nutrients was more likely to result in increased primary production. Some evidence for this contribution to primary productivity is that the absence of mussels in the Bothnian Bay, and hence a reduction in nutrient regeneration, may partially explain why pelagic primary productivity is about five times lower than in the northern Baltic proper (Kautsky and Wallentinus, 1980).

1.4.2 Biodeposition and nutrient recycling

Biodeposition is the term given to the accumulation of faeces and pseudofaeces under the mussel beds. These biodeposits may represent a significant proportion of the energy potentially available to consumer invertebrates as a food resource. Mussels may stimulate primary productivity through biodeposition (Miekle and Spencer, 1992). Nitrogen and phosphorous bound to phytoplankton and other forms of particulate matter are recycled back into the water column via biodeposition, thus reducing the immediate loss of nutrients to the sediments (Kautsky and Evans 1987). Benthic filter feeders seem to serve as important agents in stimulating bacterial growth which provides the benthos with nutritious food (Kautsky and Evans, 1987). This may be due to the fact that biodeposits, as studied by Kautsky and Evans (1987) having a C/N ratio of about 8, and so may be classified as of high nutritional value.

Kautsky and Evans (1987) found annual biodeposition by *M. edulis* to be 1.76 g dry weight, 0.33 g ash free dry weight , 0.13 g carbon, 1.7 $\times 10^{-3}$ g nitrogen and 2.6 $\times$ 10^{-4} g phosphorous per g of mussel (dry weight including shells). They also found that the biodeposits had a higher organic and nitrogen content compared to controls. This may be explained by the fact that the mussels to a large extent remove and initiate the sedimentation of small particles of high organic content which would otherwise stay in suspension. They argue that *M. edulis* occupies an important role as a connecting link between pelagic and benthic ecosystems by increasing total annual deposition of carbon, nitrogen and phosphorous by 10% and by circulating and regenerating 12 and 22%, respectively, of the annual nitrogen and phosphorous demands for pelagic primary production in the area.

Biodeposition is an extremely important element in nutrient recycling as it peaks in summer when mussel activity is at its highest and food resources are most limited. At Vrangskar in Sweden, the main distribution of mussels is above the summer thermocline at about 15 m depth, therefore the nutrients are directly released into the trophogenic zone where where they are immediately available to primary production (Kautsky and Evans, 1987). Dankers and Zuidema (1995) make the point that although more nutrients are available through biodeposition, it is not clear whether speeding up of nutrient cycles actually results in increased primary production.

1.4.3 Biodeposition and the benthos

Organic enrichment of the sediment directly under the mussel culture is likely to have an immediate local impact on the biomass and biodiversity. The impact on a particular site will depend on the type of sediment, current velocity and the species present. There was no literature found on the impact of bottom culture on the benthos. This is probably because mussel beds, unlike mussel rafts, are analogous to a natural situation.

1.4.4 Competition and space

Cultured species physically compete for space with the original inhabitants. The sheer mass of relocated spat ensures success for the mussels. For every tonne of spat deposited only the same weight in cultured mussels is harvested (Lucas, 1996). Those organisms which select for the same habitat type may be out-competed. As the mussels grow they will compete for space and for food. For example, on soft substrata, sediments beneath a *M. edulis* bed can become anoxic with increased mussel cover, thereby eliminating some infaunal bivalve species such as *Mya arenaria* (Newcombe, 1935 cited in Seed and Suchanek, 1992).

As with all ecosystems, displacing some organisms invariably creates new habitats and opportunities for others. The mussel bed itself provides a habitat for a number of species, e.g. crabs (*Carcinus maenus*), gastropods (*Littorina littorea*), and the polychaete (*Neris virens*). Deposit feeding worms profit from the organic matter that is deposited as pseudofaeces. In addition, some associated species can be considered an integral part of a mussel bed because the bed provides a hard substrate for attachment e.g. barnacles, hydroids and seaweeds (Seed and Suchanek, 1992). As *Mytilus* beds age and grow, they increase not only their biological component, the living mussels, but they also enlarge their physical component, producing structurally complex entities that are capable of harbouring a diverse assemblage of associated fauna and flora (Seed and Suchanek, 1992). The interconnection of byssal threads provides numerous interstitial habitats for a myriad of associated fauna (Seed and Suchanek, 1992).

In a natural situation species diversity and biomass of deposit feeders and meiofauna are generally higher in mussel beds than on adjacent bottoms without mussels. However, in a mussel culturing area water exchange may be limited and so the situation may be quite different (Kautsky and Evans, 1987).

From the point of view of nature conservation mussel beds have an intrinsic value of their own. It has been shown that the community metabolism of a mussel bed is significantly different from the metabolism calculated on the basis of individual mussels. Like other congregates of living organisms such as oyster beds or coral reefs they influence each other and create new habitats for other species or increase survival chances for themselves (Dankers and Zuidema, 1995). However, where harvest time is just 9-18 months after sowing it is unlikely that the cultured mussel community gains this complexity, and if it does, the destruction of such a habitat is another factor worth considering.

REGIONAL EFFECTS

Mussel farming may be responsible for some far-reaching impacts. These impacts shall be discussed under the heading regional effects, which in turn is divided into the following four sub-headings.
1.4.5 Competition for phytoplankton.
1.4.6 Knock-on effects on predators.
1.4.7 Transfers of organisms.

1.4.5 Competition for phytoplankton

One of the greatest potential impacts of filter feeder cultivation is the net loss of energy, in the form of phytoplankton, from the ecosystem. Folke and Kautsky (1989) estimated that a mussel culture farm, on the west coast of Sweden, would require a minimum area that was 20 times its own surface area to be supplied with phytoplankton. The effect of this depletion will vary hugely depending on nutrient inputs, flushing times, season and dependent biomass. In the western Wadden Sea mussel culture occupies an area of 35 km^2 (Dankers and Zuidema, 1995) of a total of 1059 km^2 (van der Veer, 1989) i.e. 3.3% . The average sized population of mussels in the Wadden Sea can pump the equivalent of the contents of the Western Wadden Sea through their gills within one week, removing almost all suspended organic and inorganic matter (Dankers and Zuidema, 1995). Asmus and Asmus (1991) measured a 37 +/- 20% reduction in phytoplankton biomass, between the inflow and outflow of the flume, over a mussel bed in the Wadden sea. Van der Veer (1989) calculated that the annual food demand for the biomass of the western Wadden Sea was 100 g ash free dry weight (AFDW) m^{-2} and of this 26 g AFDW m^{-2} is withdrawn by mussel culture. This calculation demonstrates that mussel culture may well have an impact and can be considered a serious food competitor.

This potentially large impact of mussel culture on the ecosystem could conflict with other important functions of the system, such as being a breeding area for birds and a nursery area for fish. Mussels serve as an important food source for a wide range of organisms (e.g. starfish, eider ducks and oystercatchers) (Dankers and Zuidema, 1995). Kautsky and Evans (1987) calculated that for waters of low turbulence in the

Baltic sea, that smaller particles (<18 μm size) have up to 7 times higher chance of being filtered by mussels than larger ones which sediment out quickly. This finding conflicts with that of the Nature Conservancy Council (1989) which found that mussels prefer larger phytoplankton. Either way, selection for one size only could lead to an overall shift towards one size phytoplankton in the water column. Due to competition for resources this could have a knock-on effect on the zooplankton (Rodhouse and Roden, 1987) and indigenous filter feeders in the region.

Increased competition for phytoplankton may result in a shift from worms, and small Mollusca towards large mussels. This will undoubtedly influence the areas value for other species (Dankers and Zuidema, 1995). In the western Wadden Sea the introduction of mussel culture between 1950 and 1960 resulted in a large increase in biomass in the subtidal and has increased competition for food resulting in a reduction of the intertidal macrofauna (van der Veer, 1989). If the biomass of the intertidal is very large or there is a large population dependent on it the knock-on effects may be dramatic for intertidal species and ultimately for predators. For example, Zwarts (1991) observed that 25% of the wading birds in Wadden Sea were feeding on mussel beds that only covered 3% of the area of the intertidal flats.

Haamer (1995) suggested that these filtering properties of mussels could be utilised to control eutrophication. Certainly in the western Dutch Wadden Sea eutrophication has benefited mussels as they have increased in biomass both subtidally and intertidally (Van der Veer, 1989; Beukema, 1989). Likewise, their presence has probably prevented eutrophication becoming a problem in this area.

There is considerable concern that mussel cultivation will permanently remove nutrients from the ecosystem, yet of the energy consumed only a proportion is removed from the system at harvest. Mussels filter particulate organic matter out of the water column, of which, approximately 80% is assimilated and 20% is voided as faecal waste and pseudofaeces (Rosenberg and Loo 1983). In fact, Rodhouse *et al*. (1985) estimated that 28% is returned to the ecosystem as faeces, 57% is used in metabolism and only 15% is finally harvested.

1.4.6 Knock-on effects on predators

Mussel culture means that there is more food for eider ducks, starfish (*Asterias rubens*) (Dankers and Zuidema, 1995) crabs (*Carcinus maenus*) and other predators in the subtidal zone. This brings these organisms into conflict with the mussel farmers. Of the bird species eider ducks and scoters seem to cause the most concern. Farmers in Canada have likened their farms to unplanned duck enhancement projects (Pirquet, 1990). Scottish farmers use a combination of netting and scaring devices to protect their crop (Edwards, 1995) with varying degrees of success. Milne and

Galbraith (1986, cited in Meikle and Spencer, 1989) looked in detail at the situation in Scotland, highlighting the problem and suggesting management techniques for the protection of mussel farms from these predators. In Ireland, eiders are relatively rare and so there is little threat from this source to mussel farms.

On the other hand some bird populations have been adversely affected by mussel cultivation. For example, in the Dutch Wadden Sea, over-exploitation of intertidal mussel and cockle beds and bad spatfall of both mussels and cockles since 1988 has had a negative impact on bird populations. Here mussel beds and cockle beds (subtidal and intertidal) serve as an important food source for a range of organisms, either directly or indirectly, by providing shelter and creating space for associated organisms that consume each other or are consumed by visitors such as a variety of bird species (Kamermans, 1993; Dankers and Zuidema, 1995). Overfishing of mussels and cockles, coupled with poor recruitment resulted in oystercatcher (*Haematopus ostralegus*) showing low reproductive success and eider ducks (*Somateria mollissima*) either starved or died because of parasitic infections having moved to crabs as an alternative food source (Dankers, 1993; Lucus, 1996).

Because mature mussel beds need many years to develop, it is feared that the loss of these beds through overfishing and poor spatfall has impoverished the Wadden Sea for a long time into the future (Dankers and Zuidema, 1995), and may ultimately affect ecology of this international nature reserve. Birds are regularly monitored and any change in their numbers is immediately noticeable. It is likely that this reduction in spatfall will also have an impact on the populations of crabs and starfish.

Movement of spat may have negative consequences for the predators in the source area and produce a positive impact on the predators in the receiving area. For example, knot feed on mussel spat in the intertidal zone (Cramp and Simmons, 1977) and crabs and starfish are the main predators of mussels and farmers regularly clear them from their plots to enhance mussel survival (van der Veer, 1989).

1.4.7 Transfers of organisms

In aquaculture, organisms are frequently moved from one location to another to enhance profitability. As *Mytilus edulis* is a native species any impacts relating to transfers would be on a national rather than an international level. There are three basic concerns associated with organism transfer; recruitment, genetic impacts and disease.

Recruitment

Dredging of several thousands of hectares for mussel seed must have some impact on the natural system but this assumption is unsubstantiated in the literature. The majority of seed is dredged from subtidal beds and mussels are moved to areas where they will have a better chance of survival (Hickman, 1992). About 15-20% of the necessary seed are dredged from intertidal beds (Dankers and Zuidema, 1995), therefore these beds do not get a chance to develop into mature mussel beds. Because mature mussel beds suffer damage both from human impact and natural causes and even disappear occasionally, undisturbed spatfall is essential for mature beds to regenerate (Dankers and Zuidema, 1995). Thus this movement of spat could have long-term effects on mussel bed regeneration over a wide area.

Evidence to date indicates that spat collection is not significantly affecting recruitment in wild shellfish populations in Scotland and that recruitment is largely influenced by other factors (ICES, 1994). This situation must be monitored to ensure that this continues to be the case.

Genetic impacts

Any genetic dilution should be avoided as this could theoretically lead to a weakening of the species and ultimately a drop in survival. If hatchery production of seed for commercial mussel farming were to become a major source of supply for the various mussel industries there would be a requirement for further research into its genetic implications. In addition, care should be taken, insofar as possible, to avoid moving mussels further than is absolutely necessary as this may cause dilution of any unique genetic traits in the local population.

Disease

Disease is not documented as a problem in mussel cultivation. However, care should taken to avoid introducing mussels carrying disease or parasites into an uninfected area. Care must also be taken that the mussel does not inadvertently act as a carrier of disease of some other organisms.

1.5 Suspended mussel cultivation

For suspended culture, mussel spat is collected either directly from the water by larval settlement on spat ropes or collectors, or is scraped from the rocks during spring or early summer. This may be on-grown in several ways depending on how it was collected. If the spat is collected on pegged nylon ropes, the mussels can grow *in situ* to full size. The pegs serve to increase surface area and to prevent the mussels sliding off as their weight increases. If the spat is collected from rocks, it is fed into mesh stockings. These are also used to hold the thinnings from overloaded ropes. These stockings are usually wrapped around a conventional mussel rope for support and the mussels grow in place until they reach market size. These mussel culture support structures are suspended in the water from either longlines or rafts in the on-growing areas (BIM, 1982).

Mussel rafts are usually based around a catamaran design. They consist of a set of beams strung across two flotation hulls. Attached to these beams are the mussel ropes which hang down into the water. The make of these structures varies widely from a homemade timber and expanded polystyrene type, capable of supporting 2 tonnes of mussels, up to large glassfibre platforms capable of supporting 100 tonnes (BIM ,1982). In Ireland, longlines are a more popular method than rafts (Wall, 1993). Longlines consist of flotation barrels which are used to support a stout double headrope from which the mussel ropes (or stockings) are suspended. Unlike rafts, longlines are open to the sea on all sides and thus a better overall growth rate is achieved (BIM, 1982a). This has resulted in their increasing use over recent years (Wall, 1993). Both longlines and rafts are used for intensive mussel cultivation, particularly on the west coast, e.g. in Killary Harbour (Co. Mayo) and Bantry Bay (Co. Cork). The production of rope grown mussels is increasing steadily and is estimated to reach over 7, 500 tonnes by 1996 (BIM, 1992).

1.6 Impacts of suspended mussel cultivation

The impacts of suspended mussel cultivation are very similar in many ways to that of bottom culture. The local impacts tend to be different whereas the regional effects tend to be similar. This is because local impacts depend more on cultivation method and regional effects depend more on species type.

LOCALISED EFFECTS

Localised effects will be looked at under four subheadings:
1.6.1 Water composition.
1.6.2 Biodeposition and the sediment.

1.6.3 Biodeposition and the benthos.
1.6.4 Competition for space.

1.6.1 Water composition

Chemical changes will be similar to that of bottom culture. Although it is known that mussel beds act together, it is not known if this is also true for mussels held on longlines. This may change the impact in some way. Changes in water quality have been detected in water passing through a shellfish farm, with both ammoniacal nitrogen and inorganic phosphorous levels increasing (Meikle and Spencer, 1992). There are reports of large ranges of fluxes for many of the same nutrients both within the same study site and among sites, therefore, the impacts of shellfish culture can be difficult to quantify (Dame and Danker, 1988; Hatcher *et al*., 1994). A study in Nova Scotia, Canada on suspended mussel culture showed that ammonium efflux at the mussel-line site was 10 times higher than at the reference site during the summer period. Ammonium reached a maximum of 16.15 mmol m^{-2} d^{-1} at the mussel line site (Hatcher *et al*., 1994). This compares favourably to 62 mmol m^{-2} d^{-1} measured under a salmon farm (Hargrave et al 1993). Hatcher *et al.* (1994) noted that in mussel farms in Nova Scotia the sediments of the reference site were a net sink for total dissolved nitrogen, while the sediments under the mussel line were a source.

1.6.2 Biodeposition and the sediment

Whereas bottom culture of mussels mimics a natural wild bed, suspended culture has no parallel in nature. The numbers of mussels depositing on an area of the seabed is likely to be far greater than in nature and unlike bottom culture which ideally requires a well flushed gravelly substrate, suspended culture may be practised over any substrate type.

The issue of most concern regarding biodeposition is the intense concentration over in a limited area. Before cultivation zooplankton grazing occurs throughout the system and associated processes, such as excretion of ammonia and deposition of faecal deposits are widespread. By replacing zooplankton grazing with mussel grazing these processes will be concentrated rather than dispersed. This might be expected to alter the composition and distribution of benthic fauna (Rodhouse and Roden, 1987; Meikle and Spencer, 1992).

Biodeposition affects the sediments and benthic communities to a degree that varies widely between sites and appears to be related to current velocity (Fischer, 1994). The most vulnerable areas are those with slow currents and shallow waters. Dahlback and Gunnarsson (1981) examined the environmental impacts of intense mussel farming in Sweden, where the currents are generally weak (3 cm/s). The sedimentation rate under the culture (3 g $C/m^2/d$) was three times higher than at the nearby control site. In contrast, Kaspar *et al.* (1985) working on water quality at a green lipped mussel farm with current velocities up to 110 cm/s found no significant differences between inorganic and organic nitrogen, total and soluble reactive phosphorous, silicate, calcium and magnesium concentrations in the centre of the farm and at a control site. This is in agreement with Rodhouse *et al*. (1985) who reported well-dispersed biodeposits from the mussel rafts in Killary Harbour.

Grant *et al*. (1995), in a study in Nova Scotia found that the net sedimentation rates were 1 cm yr^{-1} for the reference site and 2.3 cm yr^{-1} at the mussel line site. According to Hatcher *et al.* (1994), who investigated the same site, sedimentation rates were always significantly higher, by at least a factor of two, at the mussel-line site compared to the reference site. However, he also noted that traps tended to overestimate sediment and that most of the trapped material may be exported rather than incorporated in the sediments.

If biodeposits accumulate, this may result in increased oxygen consumption, anoxia and denitrification (Kaspar *et al*., 1985)as well as increased sulphate reduction (Dahlback and Gunnarsson, 1981). The sediments under mussel farms will become enriched with carbon, nitrogen and phosphorous (to a lesser extent). This enrichment has been reported to change the characteristics of the sediment under farms (Dahlback and Gunnarsson, 1981; Kasper *et al*., 1985). They found that the sediment under mussel cultures had a finer texture, lower bulk density and a higher water content than those at adjacent stations. Mattsson and Linden (1983) also found sediments under mussel farms to be slightly finer and in addition noted that they had a higher organic content and a negative redox potential when compared to reference sites.

As a result of biodeposition the oxygen consumption of heterotrophic organisms in the sediments will increase. Jorgensen (1980) reported that mussel beds in Denmark increase the benthic respiration per m^2 ten fold and thereby enhance oxygen depletion of the bottom water. When the oxygen demand exceeds the available oxygen then the redox potential decreases and the sediments become anoxic (Meikle and Spencer, 1992). As the sediments become anoxic a build up and release of hydrogen sulphide, ammonium and methane may result (Dahlback and Gunnerson, 1981). Hatcher *et al*. (1994) measured concentrations of 10 ppm of hydrogen sulphide at the mussel line site 6 cm below the sediment surface, rising to 196 ppm at 44 cm depth. In contrast, reference site concentrations of hydrogen sulphide were not measurable until a depth of 30 cm (2 ppm) which rose to 41 ppm at 40 cm depth. In this situation outgassing of hydrogen sulphide can happen. If this happens local populations of fish or other organisms may be adversely affected, although there is no evidence of it causing harm to mussels. In well oxygenated waters hydrogen sulphate is rapidly converted to harmless sulphate and therefore if a farm is located in well flushed waters anoxia and outgassing should not be a problem.

Sedimentation beneath the farms will also be due to the presence of artifical structures within the water body which provides an impediment to the flow (Kirby, 1994b). Anything which slows the flow of water will cause it to drop part of its sediment load therefore increasing the amount of sedimentation. The same principle will apply to trestles, cages, longlines and rafts.

1.6.3 Biodeposition and the benthos

Pocklington *et al*., (1994) looked at the polychaete response to different aquaculture activities at several sites in Canada. The polychaetes which dominated the fauna beneath the mussel lines were different from those beneath the fish cages. In both sites the sediments beneath the shellfish lines were black, finely pelleted and had high organic content with *Nephtys neotena* the dominant macrofaunal organism. Increased benthic microbial activity will often result in oxygen depletion and low macrofauna diversity as shown by Mattsson and Linden (1983) and Kaspar *et al.* (1985). According to the FAO (1992), depletion of dissolved oxygen in the interstitial waters of organically enriched sediments results in the mortality or emigration of most species characteristic of undisturbed sediments. In addition, changes in algal (epibenthic as well as planktonic) production and/or species composition may result if the ratio of nitrogen to phosphorous is altered by the presence of mussel-lines.

The species composition may alter and this often results in a higher biomass under the mussel lines due to enrichment. The biomass of macrofauna under mussel lines in Nova Scotia was noted as higher than the reference sites at all sampling times. The most significant differences observed were higher abundances of gastropods at the mussel line site, particularly in the autumn and higher abundances of polychaetes than at the reference site (Grant *et al.*, 1995). In addition, Hatcher *et al*. (1994) reported that the average weight of macrofauna was always higher for the individual at the mussel-line site.

In 1983, Mattsson and Linden examined the changes in bottom fauna under longlines of *Mytilus edulis*, as a result of the accumulation of mussel mud in Sweden. Originally the dominant benthic fauna were *Nucula nitidosa*, *Echinocardium cordatum* and *Ophiura* spp.. Three months after the culture was set up the benthic fauna was still quite similar. As time progressed the faunal composition gradually changed and was replaced by other species, better fitted for the changed conditions (higher organic sediments and a negative redox potential). The first species to disappear were the brittle stars *Ophiura* spp., after 6 months of cultivation. Other species decreased successively, *E. cordatum* after 11 months and *N. nitidosa* after 15 months. The original fauna were replaced by opportunistic polychaetes, such as *Capitella* *capitata*, *Scolelepis fuliginosa* and *Microphthalmus sczelkowii.* Shannon-Weiner diversity indices were calculated before and during cultivation. The index dropped from 4.1 to 1.6 after 15 months of cultivation, and thus were indicative of pollution. In summary, within 6-15 months of establishing a mussel farm, bivalves, sea urchins and brittle stars were replaced by opportunistic polychaetes. For this reason care should be taken to avoid siting shellfish farms over seabed communities of known conservation interest and value (Mattsson and Linden, 1983).

Dahlback and Gunnarsson (1981) also observed a change in the benthic organisms when they studied a site in western Sweden where mussels were cultured at a low current velocity in shallow water. They found that mussel faeces deposited on the seabed resulted in colonisation with a sulphur bacteria *Beggiatoa* spp.. In contrast, Rodhouse *et al.* (1985) found no detectable deleterious impacts on macrobenthos in fast flushing sites in Ireland and Scotland.

Recovery of Sediment and Fauna

A very important issue relating to all impacts is how long site recovery takes once the aquaculture products are harvested. In order to address this question Mattsson and Linden (1983) studied the recovery of their longline site post-harvest. Six months after harvest the bottom was still covered with a black anoxic hydrogen sulphide-rich mud, and the benthic fauna was still dominated by opportunistic species. One year after harvesting, the fauna was dominated by the same species but the number of species had increased due to some newly settled juveniles. A year and a half later, the organic content was still high but the redox potential had increased. The faunal community remained affected with the opportunistic polychaetes dominating. The rate of recovery is largely site specific, depending on factors such as water movement to speed up the recovery process. Fallowing is certainly one option with trawling of old sites to assist oxygenation and mineralisation of wastes (Burnell, 1995).

REGIONAL IMPACTS

The regional impacts are more related to the species identity than the cultivation method. Therefore, many of these impacts will be the same or similar to bottom cultivation of *Mytilus edulis*. Where data existed separately for suspended culture in the literature it is presented here. The subheadings as follows are the same as for bottom cultivation;

1.6.4 Competition for phytoplankton
1.6.5 Knock -on effects on predators
1.6.6 Transfers of organisms

1.6.4 Competition for phytoplankton

Mussels, oysters, clams and scallops feed by filtering phytoplankton and other particulate material out of the water column. The high filtering rates of shellfish may result in the depletion of essential nutrients (nitrogen, phosphorous and carbon) from coastal waters (Miekle and Spencer, 1992). This results in a net removal of food from other filter feeders (FAO, 1992). The amount removed will depend on the size of farming operation, farm siting, and environmental conditions. This loss may alter the ecosystem structure and may affect the recruitment of other commercially important marine species (Folke and Kautsky, 1989). For example, the trophic web in the Ria de Arosa, Spain (Freire *et al*., 1990) has been altered considerably by the introduction of intense mussel culture *Mytilus edulis* culture on some 2000 rafts comprising 10% of the surface of the ria. Because of the high filtering capacities of mussels sustenance of maximum growth of farmed mussels could require phytoplankton from an area many times the size of the farm. A depletion of phytoplankton in an area may affect the food available to other organisms and ultimately to birds.

A study in Killary Harbour, Ireland, showed that when over half the primary productivity is diverted to mussel rearing severe modifications of the environment and decreasing yields per unit area may be expected (Rodhouse and Roden, 1987). The high filtering rates of shellfish may result in the depletion of essential nutrients (nitrogen, phosphorous and carbon) and dissolved oxygen from coastal waters. With each tonne of mussel produced approximately 0.5 kg of phosphorous, 6.6 kg of nitrogen and 32.5 kg of carbon are removed from the system (Rodhouse *et al*., 1985). At high densities mussels are able to deplete the water mass of plankton, leading to nutrient depletion in mussel culture areas. Kaspar *et al.* (1985) found this for the green lipped mussel in New Zealand.

Phytoplankton depletion will not only affect other organisms in the ecosystem but if the carrying capacity is exceeded this may erode the profitability of the mussel farm itself. An example close to home is Bantry Bay where 3,000 tonnes of long-line mussels are grown in a few very restricted sites and growth rates have reportedly dropped by about 25% over the last five years (Burnell, 1995). On the other hand suspended culture may be less affected by the drop in phytoplankton levels as they may be well placed to filter out what enters the ecosystem and little may be left for other filter feeders in the intertidal. This difference is exaggerated by the fact that subtidal mussels feed continuously whereas those wild mussels in the intertidal zone can only feed for part of the tidal cycle.

There is a four-fold difference between the production rates of raft cultured *M. edulis* and wild subtidal mussels from the west coast of Ireland (Rodhouse *et al*., 1984a).

This is because they utilise a slightly different food resource due to their location.

In some circumstances phytoplankton depletion may be desirable. We have increased the nutrient levels in our coastal waters through sewage discharge, fertiliser run-off and dumping at sea. In some cases this has resulted in eutrophication of the sea, as is seen in Dublin Bay every autumn. Haamer (1996) looked at suspended mussel farming as a tool to improve water quality in a eutrophic fjord. It was shown that mussel farms covering just 1% of the fjord with rafts (or 2.4 % with longlines) would lower surface water dissolved inorganic nitrogen levels by 20%. The biochemical oxygen demand (BOD) in the deep water could be lowered by up to 26%. Decreased BOD could imply improved survival opportunities for higher life in these waters. It is also shown that if mussel farms are located so that faeces from the farms accumulate for more than a decade, the net effect of the farms will be increased BOD in the basin water (Haamer, 1996). Haamer concludes that the impact of mussel culture on the ecosystem appears to be positive even when the sediments are not dredged up. However, while this may be true for this particular ecosystem from the evidence presented it certainly is not true for all.

1.6.5 Knock-on effects on predators

Cultivation of mussels on long lines or on rafts will result in a slightly altered habitat on the seabed which some organisms can use to their advantage. Such an organism is the crab (*Liocarcinus arcuatus*). Under suspended cultivation of mussels the diet of *Liocarcinus arcuatus* changes from one composed mainly of Crustaceans (40.0%) and seaweeds (30.9%) to one consisting of mussels (43.5%), *Zostera* (15.8%) and epifauna such as *Pisidia longicornis* (10.0%) (Freire *et al*., 1990). In fact populations of several invertebrate predators such as the starfish *Asterias rubens* and various crab species have been found to increase under mussel farms as a result of mussels dropping down onto the sediment (Earll *et al*., 1984). Diving birds may also benefit from this extra food resource. Unlike bottom cultivation, the suspended culture crop can be protected from diving birds using scaring devices or underwater anti-predator nets. This is an advantage of this method of cultivation to mussel farmers. Scaring devices are being used in Nova Scotia where the problem is particularly acute. For example, in 1994 a flock of less that 100 scoter took 75% of the mussel socks on a 40 acre farm. In this situation a large amount of the losses are due to mussels being knocked off rather than consumed (Day, 1995).

1.6.6 Transfers of organisms

See bottom culture points 2 (genetic impacts) and 3 (disease) page 37.

Summary of ecological impacts of mussel cultivation

1. IMPACTS ASSOCIATED WITH BOTTOM CULTURE

• Benthic enrichment due to biodeposition may result in a change in community structure beneath the farm.
• Cultured shellfish may compete for space resulting in the original habitats being suffocated as they attain full size.
• Shellfish beds can deplete phytoplankton biomass resulting in food shortage of other filter feeders and ultimately for predators near the top of the food chain.
• A mature shellfish bed may add to the biodiversity of an area. However, this advantage may be counterbalanced by the loss of such a community at harvest.
• Shellfish cultivation will usually have an effect on water chemistry. How persistent this is will depend on the size of the culture area and the flushing time.
• Shellfish can play an important role in nutrient cycling via biodeposition, thus contributing to the nutrients available in the ecosystem.
• Cultured shellfish recycle a significant amount of nutrients making more nutrients available for primary production and potentially counterbalancing the extraction of phytoplankton from the system. However, to what degree this occurs is unknown.
• Predators numbers are likely to suffer as they are routinely removed from the beds by the shellfish farmers.

NOTE;
The impacts of harvesting bottom cultured mussels are discussed under the section on dredging (page 59).

2. ADDITIONAL IMPACTS ASSOCIATED WITH SUSPENDED CULTURE

• Biodeposits generally accumulate in sites which are not well flushed.
• Accumulated biodeposits can result in a change in the sediment characteristics with sediments becoming finer, enriched and in some cases anoxic.
• Benthic enrichment due to biodeposition can change the community structure under the farms dramatically. Recovery can take years. The severity of such an impact will depend on the size of the area under cultivation and the hydrology of the area.
• Suspended shellfish farms lose a proportion of the crop to the seabed, thus enhancing the area for predators.

SCALLOP (*PECTEN MAXIMUS*)

1.1 Method of scallop cultivation in Ireland

The only scallop cultured in Ireland at present is the native scallop (*Pecten maximus*). Trials with the Japanese scallop (*Patinopectin yessoensis*) in Wexford were unsuccessful (D. Minchin, pers. comm.). Spat is collected in Mulroy Bay, Co. Donegal or Bantry Bay, Co. Cork (Slater, 1989; Minchin, 1995). The normal method of collection is to use a small mesh bag containing a small piece of fishing net. In nature the spat settle, grow to 6-10 mm, detach, and fall to the seabed. In this system the outer mesh bag retains all detached spat (Slater, 1989). Following collection, the scallop spat is placed into either plastic trays or in pearl nets. These are primarily designed to cater for scallop growth up to 15 mm. On reaching this size, the spat are transferred to lantern nets. Low salinity areas are unsuitable for cultivation as scallops grow best at salinities greater than 30 ppt (Berry and Burnell, 1981).

Lantern nets are usually used for growing the scallops at the juvenile stage, i.e. after growth in the pearl nets and before relaying on the seabed. They may also be used to grow the scallops to full market size even from the spat stage. An alternative is the cheaper method of ear-hanging. This involves attaching the scallops to a line by means of a wire or plastic tag passed through a hole drilled in the ear of the shell. Plastic trays, lantern nets, pearl nets and ear-hung scallops are held suspended from subsurface longlines (Berry and Burnell, 1981). Subsurface longlines enable scallops to be grown at depth, undisturbed by wave action (Berry, 1981). This is essential for good scallop growth. Once the spat reach 35-40 mm, growth can be completed in lantern nets, ear-hanging or reseeding the scallops in selected areas of the seabed. If the grower goes for the latter option, only 50% of the scallops will survive. This option involves maintenance of the seabed with regular brushing to remove starfish (Berry and Burnell, 1981). Scallops are cultured in Dunmanus Bay, Co. Cork, and Killary Harbour and Kilkiernan Bay, Co. Galway (La Tene, 1995). Scallops are normally harvested at 3-5 years of age at a minimum shell length of 100 mm (Minchin, 1995).

1.2 Data available

There is a dearth of information on the ecological impacts of scallop cultivation. The majority of the information available relates to harvesting natural scallop beds by dredging. As this is the method used for harvesting cultivated beds this literature is reviewed and the impacts are assumed to be similar. As to the actual impact of collecting, sowing and growing scallops again there is little information. However, because scallops are filter feeders, it is assumed that they will have a similar environmental impact to mussels in suspended cultivation.

The ecological impacts of scallop cultivation will depend on which culture method is chosen. The two options practised in Ireland are:
a. Suspended culture in lantern nets, followed by ear-hanging or continuation in lantern nets to market size.
b. Suspended culture followed by ongrowing from 40mm to market size on the seabed. The impacts of both methods are considered below. It is been decided that, given the paucity of information on scallop cultivation, both cultivation methods would be dealt with together.

1.3 The impacts of scallop cultivation

The impacts will be looked at, insofar as is possible, from a local and a regional point of view.

LOCALISED EFFECTS

The localised impacts will be dealt with under the following sub-headings;
1.3.1 Water composition.
1.3.2 Biodeposition, the sediment and the benthos.
1.3.3 Competition for space.

1.3.1 Water composition

No information has been found on the effects of scallop cultivation on water composition. It is assumed that as they are filter feeders, like mussels, changes in water composition will be similar due to scallop cultivation. The impacts on water composition are likely to be transient and localised.

1.3.2 Biodeposition, the sediment and the benthos

Again no information has been found on the effect of scallop cultivation on biodeposition. It is assumed that as they are filter feeders, like mussels, changes in the sediment and benthic organisms will be similar. The main impacts are likely to be increased sedimentation related to current velocity, followed by a change in fauna to one typical of an enriched sediment.

1.3.3 Competition for space

The scallops will compete for space with the original inhabitants of the benthos. Bottom cultivation involves planting of thousands of half-grown scallops on the seabed. This may cause a major change in the seabed substratum, may smother existing flora and fauna and cause increased competition between organisms. It may make the habitat more suitable for some species but in the short term is likely to result in a decrease in species diversity. In the longer term a mature scallop bed will probably contribute significantly to the biodiversity of an area as the shells serve as a substrate for attachment by many species. As for bottom culture of mussels, regular harvesting by dredging will probably negate this effect.

REGIONAL EFFECTS

The regional impacts of scallop culture would normally deal with the those that effect the entire bay or region. These will be dealt with insofar as is possible under three headings;
1.3.4 Competition for phytoplankton.
1.3.5 Knock-on effects on predators.
1.3.6 Transfers of organisms.

1.3.4 Competition for phytoplankton

It is expected that scallop cultivation would result in increased competition for phytoplankton. As for mussel cultivation, the likely impact would be that due to excessive cultivation the ecosystem would be put under unsustainable pressure and some organisms would suffer high mortality or fail to recruit. Well managed scallop culture kept within the limits of the carrying capacity of the ecosystem should cause little or no problems. The organisms affected may be different to those affected by mussel cultivation as scallops may select a different size range of phytoplankton.

1.3.5 Knock -on effects on predators

Crabs and starfish are removed from the scallop beds by the farmers. Crabs (*Liocarcinus depuratar*) are removed from the beds in Kilkieran Bay as they are thought to be predators on the scallops (L. Sides, pers. comm.). Other impacts on predators would be expected to be similar to mussels.

1.3.6 Transfers of organisms

This section includes information on both the genetic and the disease implications of moving scallops from one area to another.

Genetic

Studies of the reproductive ecology of different populations of *Pecten maximus* indicate that there are genetically isolated stocks in different areas. This has important implications for stock assessment, for restocking and aquaculture programmes (Orensanz *et al*., 1991). Huelevan (1985 cited in Orensanz *et al*., 1991) used data from eight polymorphic loci to compare *P. maximus* populations from Scotland, Ireland and Brittany and found them to be very similar suggesting that they are all one population. However, Wilding (1996) discovered that the mitochondrial DNA from the Mulroy Bay population was very different from that of *P. maximus* from Kilkieran Bay, three locations in France, three in Scotland and two from the English Channel.

This difference in genetic material may be partly expressed in terms of shell colour variation. This may be an expression of some adaption to local conditions e.g. camouflage. Minchin (1991b) noted that brown scallops are easily overlooked by divers when attached to the brown kelp *Laminaria saccharina* Lamour. The fact that the Mulroy Bay population has been found to be distinctly different is probably due to limited planktonic movement into and out of the bay. This difference is likely to be reflected in other organisms within the bay (D. Minchin, pers. comm.). This has implications for stock maintenance. Transfers of scallop spat have been taking place since 1982 and should these continue on an extensive scale it is likely that genetic variability may become modified (Minchin, 1991b). Differences between populations reflect not only differing responses to differing local environmental cycles but also genetic adaptation on the part of local self-recruiting stocks, and hence a degree of genetic isolation between stocks (Orensanz *et al*., 1991). This uniqueness must be protected from introductions of other native species for mariculture or restocking purposes, even from within Ireland, as they could dilute the gene pool. If this genetic variation became diluted it may have the effect of weakening the whole population and reducing the probability of survival.

In this situation, where the endemic population is small and locally adapted transfers may destroy the unique phenotype of the local population, even if overall fitness is not compromised. The homogenising effect is popularly labelled genetic pollution and results in the loss of interpopulation diversity and distinct local phenotypes (Gaffney and Allen, 1992). In contrast, if the local population is not highly adapted to a changing environment then it is possible that the introduction will bring genes which may result in immediate benefits (Gaffney and Allen, 1992).

Disease

No records were found of scallops transferring disease. However, it is always a possibility, although small in Ireland as scallops and scallop spat are not imported.

1.4 Summary

Given the fact that scallops are filter feeders we would expect them to exhibit many of the same impacts as those described for mussel cultivation. Listed below are those specifically identified as being associated with scallop cultivation.

- Predators may benefit from an increased food supply. For ongrowing of scallops only 50% survival is expected (Berry and Burnell, 1981).
- If crabs and starfish are removed from the scallop beds this will impact on their numbers.
- Collecting wild spat may have a negative affect on the populations of organisms that depend on it.
- Aquaculture may introduce genotypically different species to an area thus diluting the uniqueness of local populations.

One of the major impacts associated with scallop cultivation is harvesting by dredging. Dredging of all bottom cultured species is discussed on page 59.

NATIVE OYSTER (*OSTREA EDULIS*)

The native oyster (*Ostrea edulis*) is cultured in relatively small amounts in Ireland. A firm gravel bottom is the preferred substratum and it grows best with minimum exposure to air and minimum overcrowding (Partridge, 1981; BIM, 1982). In 1977 only 5 tonnes of farmed oysters were produced in Ireland. This had increased to 100 tonnes per annum by 1986, before the introduction of *Bonamia* decimated the native oyster populations in Cork and Galway, with up to 98% mortalities (D. Hugh-Jones, pers. comm.). In 1995 farmed native production reached 397 tonnes (M. Fitzsimmons, pers. comm.).

1.1 Method of native oyster cultivation in Ireland

The usual culture method employed for the native oyster is extensive culture. This involves collection of wild spat and relaying in a more productive area (D. Clarke, pers. comm.). Native oysters spawn naturally in Irish waters where the sea temperature exceeds 16ûC for a number of weeks. This natural production varies from year to year depending on weather conditions. Attempts are now being made to produce seed in onshore spatting ponds and in hatcheries (O'Connor *et al.*, 1992). The material on which the oyster larvae will settle is called cultch. This cultch (usually mussel shells) is laid down on the seabed in spring. A layer of algae grows on the cultch, making it a suitable surface for the oyster larvae to settle on. The spat are then collected by dredging and relaid in a more productive area. Dredging is also the method of harvesting the native oyster. Oyster fisheries require some maintenance which involves removal of predators e.g. crabs and starfish (D. Clarke, pers. comm.). This type of extensive cultivation is practised in Clarinbridge oyster fishery in Galway, Tralee Bay in Co. Kerry, Clew Bay Co. Mayo and in the north channel of Cork Harbour.

1.2 Data available

There is practically no information relating to the environmental impacts of native oyster cultivation. This is probably because the method used is generally bottom cultivation and the impacts are likely to be similar to that of a natural oyster bed. The main impacts expected are the relaying of the oysters, competition for phytoplankton and dredging at harvest time although there is little or no evidence in the literature to support this.

1.3 The impacts of oyster cultivation

For consistency the impacts will be dealt with under the same headings as before i.e. localised and regional effects.

LOCALISED EFFECTS

These will be discussed under the following headings:
1.3.1 Water composition.
1.3.2 Biodeposition, the sediment and the benthos.
1.3.3 Competition for space.

1.3.1 Water composition

No information has been found on the effects of oyster cultivation on water composition. It is assumed that as oysters are filter feeders, like mussels, similar changes in water composition will occur due to their cultivation The impacts are likely to be transient due to the high dilution factor, and very localised.

1.3.1 Biodeposition, the sediment and the benthos

Again no information has been found on the affect of oyster cultivation on biodeposition. It is assumed that as, like mussels, they are filter feeders changes in the sediment and benthic organisms will be similar. The main impacts are likely to be increased sedimentation related to current velocity followed by a change in fauna to one typical of an enriched sediment.

1.3.3 Competition for space

The oysters will compete for space with the original inhabitants of the benthos. Bottom cultivation involves planting of thousands of half-grown oysters on the seabed. This may cause a major change in the seabed substratum, may smother existing flora and fauna and cause increased competition between organisms. It may make the habitat more suitable for some species but in the short term is likely to result in a decrease in species diversity. In the longer term a mature oyster bed will probably contribute significantly to the biodiversity of an area. However, as for bottom culture of mussels, regular harvesting by dredging, every 3-5 years, will probably negate this effect.

REGIONAL EFFECTS

The regional impacts of oyster culture would normally deal with those that effect the entire bay or region. These will be dealt with insofar as is possible under three headings;
1.3.4 Competition for phytoplankton.
1.3.5 Knock-on effects on predators.
1.3.6 Transfers of organisms.

1.3.4 Competition for phytoplankton

It is expected that oyster cultivation will result in increased competition for phytoplankton. As for mussel cultivation the likely impact would be that excessive cultivation would put the ecosystem under unsustainable pressure and some organisms would suffer high mortality or fail to recruit. The organisms affected may be different to those affected by mussel cultivation as oysters may select a different size range of phytoplankton.

1.3.5 Knock-on effects on predators

The impacts on predators would also be similar to mussels. The usual predators are crabs and starfish. The starfish *Asterias rubens* is responsible for serious mortalities in young oysters cultivated on the seabed. These oysters are cultivated in large plots (10-100ha) in Quiberon Bay (Barthelemy, 1991). Traditionally oyster farmers spend one to two days per week removing starfish using the traditional Faubert technique. The Faubert apparatus is made of ropes tracked on the ground and the starfish are ungripped on board. Now other methods of control are being investigated each with their own associated impacts. A specialised mechanical dredge is being developed to selectively harvest *Asterias*. Biological methods of control are also being looked into. The starfish *Luidia ciliaris*, found in deeper waters is a predator of *Asterias rubens* and steps are being taken to acclimatise this species to oyster beds. The long term effects of the traditional method is likely to be a severe reduction in the numbers of *Asterias* and an increase in their prey species. But the result of these new methods may have more serious, unexpected, knock-on effects on non-target species. Research must support these new approaches and identify other non-target organisms affected by this change (Barthelemy, 1991).

The loss of starfish to these methods must be counterbalanced by the realisation that their numbers probably reach artificially high levels in areas where oysters are laid. However, this may not reflect a true increase in their populations but a zoning in from surrounding areas (L. Sides, pers. comm.). In addition, crabs are known to be removed from plots but the method does not appear in the literature.

1.3.6 Transfers of organisms

When considering the ecological impact of transfers of organisms there are several considerations. The risks are both genetic and ecological. With regard to genetics one must consider whether the transferred organisms are capable of breeding, establishing a self sustaining population and, if so, are they likely to out-compete native species. Clearly this is not an issue with the native species *Ostrea edulis.* The main concern with this species is introduction of disease and transfer of non-target species with imports.

Probably one of the best documented disasters resulting from the transfer of organisms is the case of *Bonamia. Bonamiasis* is a disease of *Ostrea edulis* which was first described in Brittany, France, in 1979 (Meikle and Spencer 1989) where it caused serious mortalities in flat oyster stocks. It has since been recorded in the Netherlands, Spain and Ireland. Losses due to the disease are usually high, up to 80% or even higher. The organism responsible is a simple protistan, *Bonamia ostreae*. The organism has been recorded in Cork Harbour, Galway Bay and Clew Bay. In Cork mortalities of 90% were recorded in the 4-year old classes in 1986 and by the spring of 1987 significant mortalities were evident in all classes except one year olds. The Galway Bay outbreak was confined to a small inner outlet. Mortalities up to 70% were reported. Oysters from Clew Bay tested positive, but although no obvious mortality occurred, their condition was very poor. Interestingly the two areas where *Bonamia* outbreaks were most severe the numbers of oysters were high (McArdle *et al*., 1991). It is believed that the disease was originally introduced through an illegal consignment of oysters (*Ostrea edulis)* from France into the south-west of Ireland in the early 1980s (McArdle *et al*., 1991; D. Hugh-Jones, pers. comm). When the disease was recognised, French imports of live *Ostrea edulis* were banned. Considering the devastation that can occur it would be wise for managers to follow the International Council for the Exploration of the Sea Code of Practice Concerning Introductions .

1.4 Summary of the ecological impacts of oyster cultivation

Given the fact that oysters are filter feeders we would expect them to exhibit many of the same impacts as those described for mussel cultivation. Listed below are those specifically identified as being associated with oyster cultivation.

- Predators may benefit from an increased food supply.
- Starfish are physically removed from the oyster beds which impacts on their numbers.
- Imports of oysters may introduce non-target species with disastrous effects.

NOTE;
The impacts of harvesting bottom cultured oysters are discussed under the section on dredging (page 59).

MANAGEMENT OF SHELLFISH CULTIVATION

Objectives

Management objectives will change depending on the ecological importance of a site and the activities carried out on it. However, the following objectives will apply to all sites;

- To ensure that the carrying capacity of the ecosystem is not exceeded.
- The exploitation of the resource is carried out in a sustainable manner for man and for all the organisms within the ecosystem.

Strategies

One management strategy would be to follow the lead set by the Netherlands. There, the mussel industry was required to draw up a management plan to harmonise aquaculture and nature in the areas open to shellfish culture (Lucus, 1996). In addition to this plan the General Assembly of the Producers Organisation passed Mussel Seed Fishery regulations. The main points of these regulations are as follows (adapted from Lucas, 1996).

- A total allowable spat catch is decided on and broken down into individual quotas.
- A period of spat fishing is agreed.
- The closing off of areas to reserve them for the next period of spat fishing is agreed.
- The movements of participants' boats (via a black box recording system) is monitored.
- The catch landed is monitored.
- The degree of compliance is monitored and the plans also lay down procedures in case of breach of regulations (Lucus, 1996).

Following this strategy, a 1991 survey of the mussel stocks had indicated that approximately 58,000 tonnes of mussel spat were available. The amount of mussel spat fished was voluntarily restricted to 38,000 tonnes and 20,000 tonnes were reserved for the birds. The same amount was again reserved in 1992 despite only 28,000 tonnes being available. This decision was taken by the industry itself to ensure sustainability. These principles could be applied to other shellfish culture operations.

Depending on the impacts/potential impacts at the site other strategies may also be necessary. Probably the issue of most concern is carrying capacity. In order to decide whether or not to permit shellfish operations in an area it is important to consider the following;

1. The quantity of phytoplankton available in this area. This calculation will depend on the size of the bay, inputs and flushing times.
2. The current adult stock present (Goulletquer *et al*., 1994).
3. The proportion of the phytoplankton that is currently being utilised by cultivated shellfish.
4. The current ecological demand for phytoplankton and space by wild organisms.

A figure of 50% depletion of phytoplankton has been cited as a criterion of acceptable, sustainable shellfish production (Rodhouse and Roden, 1987) in order to preserve the integrity of the ecosystem. Data on the phytoplankton levels and demands of the ecosystem can be expensive to acquire but are the only way of calculating carrying capacity. Ecological modelling is one way of considering all factors specific to a given area. Models can predict carrying capacity using data such as exchange rate of water within the area, phytoplankton production, current ecological demands (birds, shellfish etc.) and inputs such as agricultural run-off, sewage works etc.. These may also enable sites to be chosen so that the impacts are minimised. The most important factors in shellfish cultivation are to have a plentiful supply of phytoplankton and a well flushed site.

Once shellfish cultivation is active in an area little can done to mitigate its impacts. However, in the case of suspended culture the sites can be fallowed and the sediments dredged to remove biodeposits (Burnell, 1995, Haamer, 1995). But perhaps the most critical management strategy is to involve shellfish farmers in maintaining not only the sustainability of their own enterprise but the ecosystem on which it depends.

Research required

In terms of this report the key research needs may be identified by the question how much culture is too much? This is practically impossible to answer as each site is subject to its own constraints and limitations. One possibility is to build a general ecosystem model that could be used for all sites. However just how useful it would be, given its generality, is doubtful (B. Ball, pers. comm.). Nonetheless, it would be better than what is available at present. Perhaps the onus could be on license applicants to supply the information to input into the model rather than pursue the route of environmental impact assessment for small projects. In addition, monitoring impacts once a farm is established is a priority. Suitable biological criteria need to be identified for this purpose.

Given the variation in the quality and quantity of information available for the species covered in this report each will be considered separately and the future research needs identified.

Mussel cultivation

As mussel cultivation is so well researched little further general research is needed on its impact. There are some questions which remain unanswered in the literature. Further research is required on the following topics:

- The impact of spat collection on predators in the collection and receiving areas
- The impact of the mussel long-lines /rafts on the flushing and sedimentation patterns in the bay.
- The impact of fallowing mussel sites and dredging them to remove biodeposits and oxygenate the sediments.
- The biological criteria suitable for use in monitoring e.g. meiofauna.
- Genetic differences in mussels at different locations and the impacts of diluting unique genetic differences.

Scallop cultivation

Little is known about the impacts of scallop cultivation (bottom and suspended). Of the impacts considered the only available information was on genetic impact. Research is needed under all the other headings.

Oyster cultivation

Even less is known about the impacts of oyster cultivation(bottom cultivation). Research is needed under all headings.

The onus for research should not only lie with the National Parks and Wildlife who have statutory responsibility for wildlife, but also with those organisations (such as Bord Iascaigh Mhara) who are committed to developing the aquaculture industry in Ireland.

DREDGING

The harvesting of mussels (*Mytilus edulis*), oysters (*Ostrea edulis*) and scallops (*Pecten maximus*), cultured extensively on the seabed, is carried out by dredging. Dredging is fairly indiscriminate in nature and the impact depends upon a large number of factors. These factors include the weight of the gear on the seabed, the type of towing gear, the nature of the bottom sediments and the strengths of tides and currents (Jones, 1992). As the impact of dredging is largely dependent on the gear type and the sediment type (Murawski and Serchuk, 1989), rather than the species being harvested, all the impacts of dredging-regardless of the species-will be dealt together.

In Ireland we use simple conventional boat towed dredges to harvest oysters, scallops and mussels (D. Clarke, pers. comm.). Conventional-type dredges, such as these, typically have a toothed bar to dig into the sediment and a steel mesh to retain the catch (Hall, 1994). The only detailed description of an Irish dredge is a scallop dredge. These are normally twin dredges 1.2 m in width with a tooth spacing of 110 mm and are towed by a vessel >10 m in length (Orensanz *et al*., 1991).

The impacts of dredging will be discussed under the following headings.
1. Sediment.
2. Geography and hydrology of the seabed.
3. Marine biota.

1. Sediment

The impact of dredging on sediment is two-fold in nature; impacts on different sediment types and the long term impact of dredging on the sediment type and structure. Any fishing gear which is towed over the seabed will disturb the sediment and the resident community to some degree, but the intensity of this disturbance is very much dependent on the details of the gear and the sediment type (Hall 1994). Black and Parry (1994) noted that dredged areas were readily distinguished from undredged, or not recently dredged areas, during the course of the dives. Their results suggest that dredges dig further into the softer rather than coarser sandier sediments. Churchill (1989 cited in Hall, 1994) estimated that coarse sand was typically penetrated to a depth of 1 cm by otter boards, where as the figures for fine sand and muddy sand were 2 cm and 4 cm respectively.

Eleftheriou and Robertson (1992) found that in a high energy environment where wave action is important enough to maintain a thorough mixing of the sediments, the disturbance from dredging does not translate into any significant change in the vertical distribution of the sediment grades. However, Langton and Robinson (1990) observed a change in the substratum from organic-silty sand to gravelly sand after dredging for the scallop (*Placopecten magellanicus*). This change was visible from pre- and post-dredge photographs. This was apparently due to the disruption of tube mats produced by the amphipod *Erichthonius* sp. Cox (1991) noted that an oyster dredge which had caused little concern on a sandy substrate caused severe damage when it was used on the softer sediments in Portsmouth as it dug to a deeper level. It ripped into both the mudflats and *Zostera* beds leaving the criss crossed furrows and shattered remains of bivalves. Anoxic and oxygenated sediments were churned up and in places the clay bedrock was brought to the surface (Cox, 1991). Parry (1996) noted that dredgers normally dig up to 6 cm into the sediments but careful fishermen remove only the surface 2 cm or less.

Black and Parry (1994) looked at sediment transport rates and disturbance. They found that natural suspended sediment levels during storms were 2-3 orders of magnitude smaller than the concentration recorded immediately behind the dredge. Some of the finest sediments may take a considerable time to settle; the time would depend on the prevailing weather and the grain size. By disturbing the fine material, dredging may cause a significant redistribution of fine sediments. This may have a permanent affect on the sediment composition. In addition, dredging may break the natural sediment bonds (cohesiveness and biological bonding) (Hall, 1994), causing increased likelihood of renewed suspension during natural storms (Black and Parry, 1994). Such changes in sediment composition may ultimately affect long-term productivity (Orensanz *et al*., 1991).

2. Geography and hydrology

Dredging also affects the geography and thereby hydrology of the seabed. It many ways it is a ploughing of the sediments and as such can cause massive disruption to the fauna as well as the sediment structure, type and appearance. Dredges shear off high hummocks, fill in low spots, changing the configuration of the bottom, removing areas more exposed to or protected from the current, exposing shellfish, worms and other shellfish dwelling species to predation (Mc Allister and Spiller, 1994). Eleftheriou and Robertson (1992) observed physical changes in bottom topography i.e. furrows instead of the natural ripples and dislodgement of stones, shells etc.

3. Marine biota

The impact of dredging on the marine biota will depend on whether the dredging was carried out before or after cultivation. The differences will be discussed under the following headings;

a. Pre-cultivation.

b. Harvesting.

a. Pre-cultivation

Dredging is used to prepare the area for the subtidal bottom laying of mussels, oysters and scallops (Berry and Burnell, 1981). Such dredging can have an adverse environmental impact. Unwanted organisms, such as starfish and crabs, are removed and the spat which is attached to cultch (mussel shell) or half grown shellfish are laid on the seabed. Any existing seabed community will be damaged by dredging and affected by the silt and mud released by the operation. This may have adverse effects on birds (Kirby *et al*., 1993). Some birds, such as long-tailed seaduck and eider (scarce in Ireland), may actually benefit from dredging as it may make more food available in the short-term.

b. Harvesting

The spread of more efficient methods of harvesting is of concern. Harvesting, by dredging, is indiscriminate in nature and can damage or destroy stocks of other food resources, such as *Zostera*, sand eels (*Ammodytes* spp.) and other molluscs (e.g. *Macoma* spp.) (Kirby *et al*., 1993). Shellfish harvesting presents some particular problems for the benthos. As shellfish are buried in the benthos, quite a significant amount of substrate must be disturbed in order to remove them. Shellfish harvesting devices, unlike conventional fishing techniques, may have long-term effects on the substrate and /or the biological assemblages of harvested patches. For example, Rothschild *et al*. (1994) showed that one of the major reasons for the decline of the Chesapeake Bay oyster population was mechanical destruction of habitat.

The effect of scallop dredging on the benthos has been investigated by several authors. Most have only considered the immediate and short-term impacts. Eleftheriou and Robertson (1992) carried out a short-term study (up to 25 dredge events) on experimental scallop dredging in a small sandy bay in Scotland in order to quantify the effects on the benthic fauna and on the physical environment. They observed that sessile forms such as polychaetes showed a noticeable decrease, and the burrowing spatangid *Echinocardium* was substantially reduced from the dredged area. Very large concentrations of the burrowing sand eel *Ammodytes* were also destroyed. Large numbers of molluscs (*Ensis*), echinoderms (*Asterias*) and crustaceans (*Cancer*) were killed or damaged by the dredging operation. Their overall conclusion was that the effect of dredging under these conditions was limited to the fragile and sedentary components of the infauna and the destruction of the large epifaunal and infaunal organisms.

This finding is agreement with that of Langton and Robinson (1990) and Currie and Parry (1994) who found that the species most likely to be impacted by scallop dredging are those communities associated with scallops, on or just beneath the sediment surface and which are not mobile enough to avoid the dredge. A study of dredging on a soft sediment community showed that there was a significant difference between control and dredged plots (Currie and Parry, 1994). The plot was dredged for a maximum of 3 hours per day on three consecutive days by a commercial fleet. Their study showed that scallop dredging actually changed the community structure in Port Phillip Bay, Australia. In contrast, Butcher *et al*., (1981 cited in Eleftheriou and Robertson 1992) stated that scallop dredges had few harmful effects on the marine environment.

Eleftheriou and Robertson (1992) noted that most groups of fauna which are adapted to the rigours of a high-energy environment are not affected by dredging operations in any significant way. However, the sessile fauna, the large infauna and epifauna such as molluscs, decapods, echinoderms and some polychaetes, did exhibit changes in their abundance thus highlighting their vulnerability to dredging.

Long-term faunal changes brought about by dredging have been recorded (Jones, 1992). Changes in the ecology of Port Philip Bay as noted by fisherman have been attributed (rightly or wrongly) to scallop dredging (Currie and Parry, 1994). Langton and Robinson (1990) observed that the density of *Placopecten magellanicus, Myxicola infundibulum* and *Coryphella borealis* declined between 1986 and 1987 apparently as a result of dredging for the scallop *Placopecten magelanicus*.

c. Predators

Dredging can cause high mortality in non-target species. This in turn can provide food for other organisms. For example, numerous *Asterias, Astropecten* and *Carcinus* caught between the teeth of the dredge and entrained on the bottom suffered loss of arms and limbs, or fatalities (Eleftheriou and Robertson, 1992). These mortalities provided food for fish to prey on.

Eleftheriou and Robertson (1992) observed that during their experimental scallop dredging very large concentrations of the burrowing sand eel *Ammodytes* were destroyed. This species is the staple diet of many sea birds such as arctic terns, kittiwakes, puffins, great skuas and red-throated divers (O'Connor *et al*., 1992). Hislop and Webb (1992) also reported *Ammodytes* as a component of the diet of salmon. So, dredging in an area with birds who are dependent on these species could have serious ecological impacts. On the other hand the presence of large numbers of scallops will often have a positive impact on the predator species present. Crabs (*Carcinus maenus*) and starfish (*Asterias rubens and Marthasterias glacialis*), the anemone (*Anthopleura ballii* and *Natica* spp. were all reported as predators of scallops. Fish are also suspected as benefitting from scallop spat and thus contributing to spat mortalities (Minchin 1981, 1992a).

Parry (1996) in a study in Port Phillip Bay, Australia discovered that scallop harvesting typically causes a reduction of ~20-30% in the abundance of bottom dwelling marine invertebrate animals. Some species remained at low abundance even as long as 14 months (the duration of the study) later. The abundance of fish changed only in one site and the finding was an increase of 20% in the abundance of flathead and spiny gurnard. Part of the decrease in species abundance post dredging was attributed to increased levels of predation of these species made more accessible by harvesting. Parry also noted that there was a four fold increase in snapper fish four years after intense harvesting, but could find no explanation for this. The bycatch of other species by scallop harvesting is a minor problem. However, during the intensive harvesting on experimental plots, 0.5 to 4.8 spider crabs were caught per 600 m drag and up to 55% of these were probably killed. Flounder fish were also recorded in the bycatch.

Animals living on the sediment (sponges, sea squirts) and flora (especially seagrass) are obviously vulnerable to harvesting, but none were abundant on the study sites. The relative absence of these species could be due to 30 years of scallop harvesting. Studies to develop an environmentally friendly harvester are needed(Parry 1996).

d. Recruitment

Recruitment can be affected by lack of clean, hard, growing surfaces (Minchin, 1991a), overfishing of reproductive class of adults and destruction of habitat (Visel, 1988). A possible effect on recruitment has been observed by Langton and Robinson (1990) who noted that the young-of-the-year scallops were not seen in dredged areas. In recent years efforts have been made to counteract the negative impacts of dredging e.g. laying down crushed shell to encourage spat settlement of oysters where the original material has been buried or removed by dredging (Visel, 1988; Dumbauld, 1993).

Other impacts may not be so easy to correct. Mc Loughlin *et al.* (1991) showed that 4 to 5 times as many scallops were crushed or damaged as were caught and landed by the scallop gear. This is a totally unsustainable use of the resource and will have long-term implications for recruitment and on other organisms which depend on scallops as their food source.

e. Other flora and fauna.

Dredging may also have an adverse knock-on effect on other marine biota. One concern is that dredging may destroy the amount of productive fish habitat. Associated flora and fauna are recognised as providing much more that just a food source for the target fishery. Seagrass (*Zostera*) habitat is being lost the world over and mechanical damage certainly plays its role (Kurland, 1994; Short, 1996). Seagrass not only provides food for overwintering waterfowl such as brent geese and wigeon, fish such as pollack and winter flounder feed on sand shrimp and other small organisms associated with eelgrass beds (Kurland, 1994). In addition, their dense underwater canopy has a nursery function. For example, herring and other fish lay their eggs on the surface of seagrass leaves; newly shed lobsters seek refuge there while their shell hardens; scallops settle there to avoid predation; and starfish, snails, mussels and other creatures attach themselves to the seagrass leaves. In short, seagrass beds perform critical roles as foraging shelter and nursery habitat for marine life, contributing markedly to the overall productivity of shallow coastal embayments (Castel *et al*., 1989; Kurland, 1994; Short *et al*., 1996). Furthermore, seagrasses also play a role in stabilising sediments, controlling erosion and maintaining water quality (Kurland, 1994).

4. Summary

- Dredging generally has a much greater effect on soft mud than on hard sandy sediments.
- On sandy sediment the trend seems to be loss of the older molluscs and sedentary organisms.
- On soft muddy sediment dredging changes the nature of the sediment to become more gravelly as the finer sediments get washed away in the sediment plume.
- The impact will depend on the type of dredge used.
- The habitat may take months, if not years, to recover.
- Dredging may destroy habitat and reduce juvenile recruitment.
- The impact will depend on the importance of the site e.g. if there are *Zostera* beds of importance to birds/fish.
- The community structure can be affected.
- Dredging can leave fauna open to predation as their environment is drastically changed, as they are exposed immediately after dredging.
- Overfishing can lead to population crashes in predators and ultimately in the target species itself.

5. Data available and research required

Even with research carried out to date there are inherent problems. The main problem is one of generalisation. Most of the worlds fisheries use different gear, operate on a range of substrate types and harvest scallops from different biological communities. Consequently even if the effects of scallop dredging had been investigated in several of the world fisheries, it would not be surprising if the impacts differed (Currie and Parry, 1994). Furthermore, the impacts depend on the speed of currents, type of substrate, type of dredge, amount of ground contact etc. (Jones, 1992). A lot of experiments presented lack the statistical power to detect a small impact (Peterson *et al*., 1987; Eleftheriou and Robertson, 1992) and others involve an inappropriate scale of impact i.e. areas dredged much smaller than that normally dredged in commercial operations, or site chosen not viable for commercial operations (Eleftheriou and Robertson, 1992). These factors all have implications for interpretation of results. Furthermore, a lot of studies examined before and after dredging in a regularly dredged area. This is of limited use as the ecology may have changed considerably from pre-dredge state due to the impact dredging. It would be preferable to have data before and after dredging operations in a new, large, commercially viable area.

6. Management strategies and objectives

The shellfisheries must be sustainably managed for the benefit of not only the shellfish farmer but also as an integral part of the ecosystem.

Objectives

The management objectives will change depending on the ecological importance of a site and the activities carried out on it. However, the following objective will apply to all sites:

To ensure that the exploitation of the resource is carried out in a sustainable manner for man and for all the organisms and activities within the ecosystem.

Strategies

Management strategies may include; habitat regeneration (Iribarne *et al*., 1991), restocking of depleted areas, fishing controls, imposition of minimum size limits, areal/seasonal closures and meat counts (Orensanz *et al*., 1991).

One strategy for sensitive management is harvesting by diving. In Norway a commercial scallop fishery is operated on this mode of collection (D. Minchin, per comm.). At present the threshold density for commercial diving is around 20 scallops/m^2, although it may vary as a function of economic reward (Orensanz *et al*., 1991).

SECTION 3

SHELLFISH CULTIVATION II

PACIFIC OYSTER (*CRASSOSTREA GIGAS*)

1.1 Method of cultivation

In Ireland, Pacific oysters are usually cultured intertidally on trestles. This method is practised in many small bays and on a large scale at Dungarvan Harbour, Co. Waterford and in Bannow Bay, Co. Wexford. Small amounts of Pacific oyster are cultured by bottom culture at Clarinbridge in Co. Galway (D. Clarke, pers. comm.). Intertidal culture of Pacific oysters is best suited to large, sheltered mud- or sand-flats exposed only on low water spring tides (BIM, 1983). A firm substratum is necessary to support the weight of a tractor, and to facilitate planting, maintenance and harvesting. The texture of the substratum varies greatly from site to site with medium to coarse hard sand in Dungarvan (pers. obs.), a mixture of clay and stones in Cork (D. Hugh-Jones, pers. comm.), and a mixture of sand and shell in Galway (O'Toole, 1990). Access to oysters cultured in this manner is usually restricted to 3-10 days a month at the time of the spring tides (depending on the site). This means that the time available for tending cultures is restricted. The area theoretically suitable for cultivation is between the MLWS and MLWN.

Spat is supplied by hatcheries and on-grown to market size in normally in mesh bags or less frequently in North West Plastic (NWP) trays, supported by trestles. Trestles are steel supporting structures which are normally a height of 0.5 m above the seabed (the height varies depending on exposure time). They typically have 3-4 supporting bars, 4 legs and a capacity to hold 6 bags each. Their function is to keep the oysters off the sea bottom and to prevent grit getting inside the animal. The mesh bag facilitates ease of handling and also reduces predation by crabs, starfish and birds. The mesh size of the bags is increased as the oyster grows. Maintenance consists of turning the oyster bags every spring tide. This rids the bags of any settled silt, stops growth of oyster shell into mesh, and helps destroy fouling organisms (D. Clarke, pers. comm.).

1.2 Data available

Surprisingly few studies have examined environmental changes resulting from intertidal bivalve culture (Nugues, 1996; Burnell, 1995).

1.3 Ecological impacts of cultivation of *Crassostrea gigas*

LOCALISED EFFECTS

Localised impacts will be discussed under the following headings:
1.3.1 Water composition.
1.3.2 Biodeposition and the sediment.
1.3.3 Impact on the benthos.
1.3.4 Competition for space.
1.3.5 Disturbance to birds.
1.3.6 Space occupation.
1.3.7 Knock-on effects on predators and other organisms.

1.3.1 Water composition

On an annual basis oxygen and nitrate are mainly taken up by the sediment beneath the oyster beds, where as ammonia, urea and primary amines are released to the water column. Oysters metabolic activities influence the intensity of these exchange rates by their own respiration and excretion. Soluble end products are released to the surrounding water and biodeposits modify the particulate input into the sediment.

1.3.2 Biodeposition and the sediment

Biodeposits are made up of faeces and pseudofaeces and fall onto the sediment below the trestles. These pseudofaeces consist of mineral components that the Pacific oyster rejects while sorting the seston. The faeces consists of the organic material which went through the digestive tract. The distance of 0.5 m between the trestles and the sediment may allow sufficient water movement to remove any biodeposits which may fall to the sea floor (Razet *et al*., 1990).

Organic enrichment has been recorded at some sites. Nugues *et al.* (1996) noted an increase in organic and silt composition sediment beneath the trestles. In this case water velocity was noticeably decreased by the presence of trestles which probably lead to the increase in sedimentation rate observed beneath them. Cho *et al*. (1982) found great quantities of organic matter and sulphides in the bottom mud of shellfish farms (unidentified species) in the innermost part of Jinhae Bay, Korea. These were mainly due to excrements from shellfish and fouling organisms. Other studies have shown that trestle cultivation of oysters is responsible for increased sedimentation of both organic matter and contaminants (Martin *et al*., 1991; Kirby, 1994b). Sornin *et al*. (1983) went as far as to say that the accumulation of biodeposits by oysters brings about noticeable geological modifications of the underlying sediment. He recorded an increase in the organic, silt and phaeopigment content beneath the trestles which was again probably related to the recorded decrease in current velocity at both sites (Sornin *et al*., 1983). They recorded daily deposits of 8-99 grams of carbon a square meter from directly beneath the oyster tables (Sornin *et al*., 1983).

Martin *et al*. (1991) looked at the significance of oyster biodeposition in concentrating organic matter and contaminants in the sediments. The results showed that biodeposition leads to sedimentation of matter which can reach 700 $g.m^{-2}.j^{-1}$ and 500 $g.m^{-2}.j^{-1}$ on a sandy shore and in a clay bottomed pond respectively. Sedimentation results in organic matter and chemical contaminants accumulating on the seabed. The impact was particularly noticeable in the sandy sediment, and was observed down to a depth of 25 cm. This accumulation was not irreversible. Due to the washing of sand, the vertical profiles of organic matter and contaminants in the foreshore sediment became similar to those observed in the reference sediment two months after stopping the oyster rearing and so the biodeposition.

In contrast, Cho and Park (1983) looked at eutrophication of bottom mud in Goseong - Jaran Bay, Korea, an off-bottom oyster and arkshell fishery and found no change in status since 1976. Moore (1996) investigated the impact of an intertidal oyster farm on the benthos in Dungarvan Harbour. She compared the benthos at the control site to that at the site with the oyster trestles. The sediment did not appear to be organically enriched by the farm. She concluded that her study was not carried out on a broad enough scale either temporally or spatially to justly consider organic enrichment not to have taken place. The fact that both the control and test sites were within Dungarvan Harbour, the most intensive oyster cultivation site in Ireland, means that it is possible that the whole bay may be slightly enriched. From the evidence presented it would seem that, as for the suspended cultivation of mussels, the impact on the sediment depends on the current velocity and site flushing.

In addition to physical change, chemical change was also noted. This is partly due to the recycling of the deposited organic matter which increases the oxygen needs. This provides opportunities for ammonification and reduction of sulphate into sulphur (Sornin *et al*., 1983). Nugues *et al*., (1996) also noted the increased oxygen demand by the sediment which presented itself as a reduction in the depth of the oxygenated layer beneath the trestles.

1.3.3 Impact on the benthos

Decreases in macrofaunal abundance have been detected in areas of extensive intertidal oyster cultivation (Heral *et al*., 1986; Castel *et al*., 1989). If there is organic enrichment of the sediment then there is likely to be some detectable change in the fauna. Nugues *et al*. (1996) noted small, but significant, changes in the macrofaunal community sampled beneath oyster trestles, compared with that found in adjacent uncultivated areas. These changes were associated with an increase in organic and silt composition and a reduction in the depth of the oxygenated layer of the sediment beneath the trestles.

They also noted that the main factors affecting the macrofaunal communities appeared to be linked to environmental parameters such as sedimentation rate and current velocity. In general the macrofaunal communities found in both the control and cultivated areas were impoverished and the abundance of dominant species and diversity were low. The main differences between the fauna beneath the control and the two test sites was the decreased number of spionid underneath the trestles which may have been due to increased sedimentation.

Castel *et al*. (1989) investigated the influence of oyster (*Crassostrea gigas*) parks on the abundance and biomass patterns of meio- and macrobenthos in tidal flats. Oyster parks are intertidal layings of oysters surrounded by a fence to protect them from crabs and starfish. Although we do not currently use this method of cultivation in Ireland we have done so in the past for clams and may do so in the future. Castel found that when compared to the adjacent sandbanks, oysters clearly enhanced the meiofaunal abundance (from 1130 to 4170 individuals $10cm^{-2}$) but depressed macrofaunal densities (from 640 to 370 individuals $10cm^{-2}$) . The organic rich oyster deposits probably favour meiofauna by increasing the trophic resources but do not favour macrofauna by inducing low oxygen concentrations. Moreover macrofauna are more sensitive to predation than meiofauna, probably due to size. Although *Crassostrea gigas* is a suspension feeder it does promote meiofaunal abundance. This points to the strong influence of biodeposits on structure and trophic resources available for meiofauna.

Moore (1996) looked at the impact of an intertidal oyster farm on the benthos in Dungarvan Harbour. She compared the benthos at the control site to that at the site with the oyster trestles (under the trestles and in the servicing lane between the trestles). According to the Shannon-Weiner index the fauna beneath the trestle was found to be less diverse that the control but surprisingly when Moore looked at the fauna in the lanes between trestles she found that it was more diverse than the control. She noted that the polychaete *Capitella capitata* was absent at the control site but present in the lane and trestle site. This is an opportunistic species and perhaps colonised the lane and trestle treatment site possibly because of increased food resources. Usually *C. capitata* is an indicator of organic enrichment (Dahlback and Gunnarsson, 1981) but at much higher densities than found in this study. Moore also made the point that *C. capitata* is classified by benthic ecologists as an indicator of disturbed habitats. *Nephtys hombergi* and *Tellina tenuis* occurred in higher densities at the control site than at the oyster farm. Moore suggested that differences in all three species may be due to mechanical disturbance rather than organic enrichment.

Mechanical disturbance may be due to the movement of tractors for service and maintenance. This can also lead to compression and churning up of sediments in the intertidal zone with negative effects on the invertebrate fauna (O'Briain, 1993). If plants, such as *Zostera*, are present on the foreshore, they could also be damaged by tractor activities.

Other methods of oyster culture can cause disturbance to the benthos. In Washington, USA, Pacific oysters are grown in fenced off areas, known as parks, on the ground (Simenstad and Fresh, 1995). This regime is harsh on the benthic organisms as the growers may move the oysters several times to improve their growth. Harvesting is carried out with mechanical dredges and a plot may be harrowed, dredged, raked, levelled and treated with an insecticide carbaryl to destroy burrowing shrimp several times a year. In some cases seagrass is removed to increase water flow over the plots. Moreover, activities on the most intensively cultivated intertidal plots have been repeated annually for decades. These activities impose some level of disturbance on the benthic substrate and associated community (Simenstad and Fresh 1995). This method of culturing oysters appears to cause far more environmental damage that the current trestle based method. We must resist an introduction of such a method to Ireland, particularly the practice of spraying carbaryl in the intertidal zone.

1.3.4 Competition for space

Areas which would normally be available for birds and other animals may be occupied by shellfish culture. For intertidal culture, loss of habitat can arise from the presence of structures used for growing shellfish on intertidal feeding ground. These structures include frames used for holding small spat, bags held on trestles, and areas under netting. The farming operations are generally quite small in terms of area covered (1-2 ha.). However, the cumulative reduction of feeding grounds arising from the increasing number of operations can be substantial (O'Briain, 1993).

1.3.5 Disturbance to birds

Disturbance can be defined as any situation in which a bird behaves differently from its preferred behaviour. Any overall reduction in birds feeding, as a result of this change in behaviour, may increase energy requirements, and hence adversely affect survival (Davidson and Rothwell, 1993). The main cause of disturbance will be the service and maintenance of the culture structures. Disturbance from intertidal shellfish farming is mainly caused by the presence of tractors and groups of people working on the mudflats (O' Briain, 1993). Activities on the mudflats include grading, the turning of bags on trestles, the loading of oyster bags for harvesting and the brushing of weed off clam nets (pers. obs.). On a low spring tide activity can be very intensive; 13 tractors, most with trailers and crews of four to five people, were observed on the intertidal flats at Dungarvan on a low spring tide in November 1994 (pers. obs.). The size of area disturbed will depend on the number of points of access to the strand as these will tend to concentrate or disperse the disturbance. The number of farms at the site, and their size in relation to the total area available, will determine what feeding and roosting alternatives there are for the birds within an estuary or bay. Farmers' attitudes are also a very important factor in disturbance.

Bird species, and individuals within species, vary greatly in their susceptibility to disturbance, and huge variation in reaction is seen between sites. This susceptibility is likely to vary with age, season, weather and the degree of previous exposure (Cayford, 1993). Timing of disturbance is crucial. If birds are disturbed when feeding, they are likely to move and feed elsewhere. However, disturbance to roosting birds is more likely to cause them to desert an area (J. O' Halloran, pers. comm.). Some birds use the lower intertidal zone as a roosting site at certain times of the day, and at certain stages of the tide. Shellfish farms on these areas of the shore may reduce the viability of low tide roosts. Most oyster and clam operators will have their onshore sorting operations adjacent to the shellfish farms. Tractors, trailers, sorters, trestles and boxes may be scattered over a wide area of the shoreline. These may interfere with roosting at different stages of the tide (O'Briain, 1993).

In the coastal zone, intertidal shellfish farming may be just one of many disturbing activities. Disturbance from intertidal shellfish culture affects few breeding birds. It principally impinges on wintering birds. This is because intertidal flats (mud and sand), although a minor summer habitat for breeding birds, are of major importance as a habitat for many winter visitors (Davidson and Rothwell, 1993; Coveney *et al.*, 1993). Wintering birds are particularly susceptible to disturbance. This is due to a number of factors which include the condition of the birds post-migration, limited suitable habitat, harsh weather and prey accessibility in winter. Birds arriving in Ireland from the north (Iceland, Greenland, Scandinavia and the Tundra) in winter may be in poor condition, having limited fat reserves left following their long journey. Disturbance at this point may be critical to their survival.

Heffernan (1995) showed that there is little consensus as to which birds are sensitive to disturbance. Birds vary hugely in their susceptibility to disturbance depending on location and species type. At Bull Island Special Protection Area, for example, curlews, noted as a sensitive species (Cramp and Simmons, 1977), have been seen feeding on the mudflats between the golf course and the main road (pers. obs.). Therefore, birds at every site are likely to show a slightly different response depending on the characteristics of the operation, its location, their versatility and ability to habituate to disturbance. Reaction to disturbance may initially be to take to the wing. Some birds do not move far whereas others, such as the bar-tailed godwit, dunlin, grey plover and knot, may leave the area entirely (Kirby *et al*., 1993). Such observations of these reactions are very important, because the further the birds fly, the more energy they expend and thus the greater the cost of disturbance.

Disturbance and a loss of feeding habitat to intertidal shellfish cultivation will affect a lot of species. However, disturbance will be limited to low spring tides i.e. when the area between the MLWN and the MLWS is exposed. The effect shellfish farming has on birds at a site will largely depend on the characteristics of the site and species of birds present. Every site will be different.

1.3.6 Space occupation

Space occupation is of concern as, although the extent of oyster cultivation is much less than in France, the largest cultivation areas are located within Special Protection Areas i.e. areas of conservation for birds (Bates 1995). It is well recognised that loss of habitat causes reduction in the species dependant on it. Goss-custard and Moser (1988) showed a convincing relationship between loss of habitat in the upper intertidal zone due to *Spartina anglica* invasion and a decline in dunlin numbers. Oyster farming can occupy a large amount of space on the intertidal flats and there is no reason to suppose that a similar reduction in a species dependant on the lower tidal zone could not occur. Intertidal oyster farms are located at the lower tidal levels where most shorebird species feed (Heffernan, 1995). To date we have little habitat loss to oyster farming as compared to other countries. In Marennes-Oleron Bay, France an area of 3,500 ha is licensed for oyster culture (Heral *et al*., 1986) while in Dungarvan 154 ha is licensed for the same.

Some birds are more likely to be affected by habitat loss than others. Site selection is also important; oyster farming requires a firm substrate, but not rock and so all species except the turnstone have the potential to be affected by habitat loss due to oyster farming (Heffernan, 1995). Some birds use a wide range of habitats and therefore have alternatives in the event of losing habitat in intertidal zone. The species most likely to be affected by loss of habitat are birds whose feeding and roosting habitats are suitable for shellfish farming and which feed or roost on the low shore to mid shore. Nearly all the wader species fit into this category. All species feeding on the lower shore area are likely to be affected by habitat loss to oyster farming. Ringed plover, redshank and turnstone are the only species unlikely to be affected by feeding loss due to space occupation as their prey items are at the uppermost part of the shore. Species which may lose roosting habitat to Pacific oyster culture are the golden plover as well as some geese species (Heffernan, 1995).

1.3.7 Knock-on effects on predators and other organisms

Shellfish farmers have been known to remove rocks and seaweed from the intertidal zone. This effectively limits feeding habitat for such species as turnstones (O'Briain, 1993). The turnstone is likely to be particularly sensitive to rock and seaweed removal by shellfish farming as it feeds on the organisms which live on and under seaweed and stones. This practice also affects, albeit to a lesser extent, brent geese, wigeon, teal and pintail which feed on algae which may or may not be attached to stones (Heffernan, 1995). Rocks provide a suitable substratum for mussels to attach to and crabs to shelter beneath. Rock removal could affect oystercatchers, curlews and turnstones which feed on these species (Heffernan, 1995).

Farmers have also been known to remove local mussel beds in order to prevent their spat settling on the oysters, clumping them together and clogging up the mesh of the oyster bags (O'Briain, 1993). If the local mussel beds are removed this will affect many species of birds. The most adversely affected are likely to be oystercatchers which depend on bivalves as a substantial part of their diet, and knot which feed on mussel spat (Heffernan, 1995).

REGIONAL EFFECTS

Regional effects will be discussed under the following headings:
1.3.8 Competition for phytoplankton.
1.3.9 Knock-on effects on birds.
1.3.10 Tranfers of organisms

1.3.8 Competition for phytoplankton

Competition for phytoplankton can affect wild species as the cultured species being in the majority manage to filter out the most of the phytoplankton and so the wild species relying on the same resources may suffer. Haure and Baud (1991) noted that in the bay of Bourgneuf as the stock of oysters (*C. gigas*) increased from 37,821 to 46,343 tonnes from 1986-88. The wild population of mussels dropped from 40,068 to 6,700 tonnes in the same period. They postulated that this reduction is due to trophic competition with oysters as they feed on approximately the same phytoplankton. However, there is some evidence from studies on the related species *Crassostrea virginia* that, like mussels they may be important in nutrient recycling (Dame *et al.*, 1985).

Overproduction of cultured species can result in negative feedback. For example, in 1970 when *C. gigas* was first introduced into France the oysters were harvested after two years weighing 100 g. By 1984 oysters were taking three years to reach 50 g in weight (Heral *et al.*, 1986). Efficiency of production decreases when the stock increases past a sustainable point. In Marennes-Oleron Bay the production reached a plateau maximum of 40,000 tons whereas the stock is 200,000 tons. This plateau corresponds to the maximal capacity of production of the ecosystem, limited by trophic capacities of the bay (Heral *et al.*, 1986). Such overstocking can have serious negative implications for wild filter feeders and their dependent organisms. In some areas cultivation is so extensive that it exceeds the carrying capacity of the system, resulting in poor bivalve growth and increased incidence of disease (Heral *et al.*, 1986).

1.3.9 Knock-on effects on birds

The presence of intertidal shellfish farming will mean that the birds will feed more tightly packed together at the lower tidal levels. This is likely to result in poor feeding performance because the longer the tide edge, the greater the spacing of birds and the better the feeding rate achievable by each (Evans and Dugan, 1984). Many worms and crustaceans are most active, and closest to the surface, when the tide just covers or uncovers the sediments. For this reason, many species of waders that feed upon them forage by following the edge of the ebbing and flowing tide (Evans, 1979). Perhaps the critical factor in determining the threshold for shellfish farming is not the area of intertidal flats occupied by the farm but the reduction of feeding edge at low tide level.

1.3.10 Transfers of organisms

Until 1992 Ireland had a policy of banning all mollusc importations except under licence. Following implementation of EU Council Directive 91/67/EEC, the free movement of trade in shellfish commenced in January 1993 (Minchin *et al.*, 1993). Pacific oysters are a non-native species and as such there are several potential problems associated with bringing them into Ireland. These will be dealt with under three headings:

a. Genetic implications.
b. Transfers of non-target species.
c. Ecological.

a. Genetic implications

The genetic effects of introductions on native populations may be defined as direct or indirect. Direct effects occur when the gene pool of the native population is open to the invasion of genes from the introduced population. Indirect effects occur when hybridisation between the native and the introduced population is not possible, but alterations in gene frequencies result from ecological interactions with the introduced organism. Only the indirect effects apply to Pacific oyster cultivation as they cannot hybridise with the native *Ostrea edulis* (Gaffney and Allen, 1992). The likely impact of Pacific oyster cultivation will be negligible as although it has been observed spawning in Ireland in Donegal Bay in 1993 (C. Duggan, pers. comm.), it has not been recorded as establishing wild populations. This is presumably due to the limited occurrence of sufficiently high temperatures for successful reproduction. Ecological interactions will drive genetic changes in both the native (*Ostrea edulis*) and the introduced species (*Crassostrea gigas*) (Gaffney and Allen, 1992). Global warming may enable the Pacific oyster to spawn successfully and so cause other impacts, genetic and well as ecological.

b.Transfers of non-target species

There are numerous examples of the introduction of disease and other unwanted organisms with shellfish. Perhaps the best known unwanted predator is the American whelk tingle (*Urosalpinx cinera)* introduced into England along with American oyster (*Crassostrea virginica*). This gastropod became established in some areas of Essex and Kent and caused a great deal of damage to the juvenile stages of the European flat oyster (*Ostrea edulis*) (Spencer, 1991). In British Columbia the Japanese oyster drill (*Ceratostoma inoratum*) was introduced along with the Pacific oyster (Cox, 1988). In France, introduction of the Pacific oyster led to other Japanese species living on the French coast. The source was probably species imported with the spat on collectors. The species found included the annelid *Hydroides ezoensis*, coelenterate *Aiplasia pulchella*, mollusc *Anomia chinesis*, cirripeds *Balanus amphitrite amphitrite* and *Balanus albicostatus*. The existence of Japanese species outside of the farming areas was not proven (Gruet *et al*., 1976).

Despite being an island nation, we have suffered our fair share of introductions of non-target organisms. Minchin *et al*. (1993) discovered that consignments of Pacific Oyster certified as being free of *Bonamia, Marteilia* and other species actually harboured *Mytilicola orientalis*, *Myicola ostrea*, *Crepidula fornicata*, *Ostrea edulis* and *Mytilus edulis*. The biomass of the importations and the frequency of *Mytilicola orientalis* and *Myicola ostrea* in the consignments suggest that they may become established in Irish waters. Further more recent research has confirmed that the copepod *Mytilicola orientalis* is now established in Ireland (D., Minchin, pers. comm.)

Mytilicola orientalis was not known in Irish waters until prior to the transfer of Pacific oysters from France in 1993. Introductions of Pacific oysters with *Mytilicola orientalis* must have consequences for other marine populations. This species is known to occur in various oysters, mussels, clams and trochid and other snails and to date there have been no quantified studies of its effect on these species. It is known to have taken up new molluscan hosts in British Columbia (Bernard, 1969, cited in Minchin *et al*., 1993). Infestation is likely to weaken the host, possibly leading to poor condition or death. *Mytilicola orientalis* is a potentially serious pest, not least because it is capable of transferring to the native oyster (*Ostrea edulis*) (Holmes and Minchin, 1995).

Minchin *et al*. (1993) make the point that the discovery of *Ostrea edulis* and *Mytilus edulis* in Pacific oyster consignments is worrying as they are both vectors of *Martelia refringens* and in the case of the protozoan *Bonamia ostreae*, *Ostrea edulis* is a vector. The presence of *Crepidula fornicata* also has serious implications as should it become established there may be significant changes in (a) trophic competition, (b) changes in the texture of the seabed and (c) modification of the benthos. Another species which was not detected by Minchin *et al*., (1993), but which is also high risk is

Sargassum muticum. Being a monoecious species a single plant can result in the development of a whole population and come mature within a year (Minchin *et al*., 1993).

Phytoplankton species have also been imported with live consignments of oysters. In fact, sixty-seven species of phytoplankton (43 diatoms, 22 dinoflagellates and 2 silicoflagellates) were recorded in addition to other microspecies such as foraminiferans and tintinnids. Fifteen types of dinoflagellate cysts were recorded. There is concern that potentially harmful species of phytoplankton may be imported accidentally into Ireland with shellfish transfers (O'Mahony, 1993). It is possible that a phytoplankton species, capable of causing a red tide, could be imported to Ireland in Pacific oysters from France. One "red tide", in Dungarvan in 1994, virtually wiped out all the cockles and lugworms in the area (P. Cullen, pers. comm.). This could have serious consequences for the birds that feed on these species.

Future intended introductions of exotic commercial species will continue to be subject to the full protocol of the current ICES code of practice (Minchin *et al*., 1993). However, little can be done to avoid future undesirable transfers of organisms from other EU countries.

c. Ecological

Introduced species, both larval and adult stages, occupy similar niches in the marine environment to native species, presenting a threat to the balance of existing communities. This could be a major problem, which may not be detected during pilot studies. To date, Pacific oysters have not adversely affected the indigenous fauna. This may be due primarily to its contained status (Meikle and Spencer, 1989).

1.4 Hydrology and Sedimentation

The presence of trestles has been noted to decrease water velocity causing increased sedimentation (Nugues *et al*., 1996). Their presence may have the effect of causing the water body to slow down and deposit more of its sediment load. This is certainly the case for intertidal oyster and mussel farms in France where about 30% face problems of sedimentation. This problem forces occasional relocation and abandonment of the beds (Weston, 1990). Kirby (1994b) discusses ways in which the presence of trestles can result in increased sedimentation and ways in which this can be avoided. Not only can this increased sedimentation have a negative affect on the farm itself, with records of whole oyster farms being destroed by smothering (Kirby, 1994b), but this could also negatively affect the infauna and subsequently the bird populations that feed on them (Weston, 1990).

1.5 Summary

In summary, the impacts, probable and actual, of intertidal Pacific oyster cultivation are:

1. Increased ammonia concentration in the water column.
2. Organic enrichment beneath the trestles.
3. Reduced current velocity and thus increased sedimentation.due to the presence of trestles.
4. Disturbance or compression of the sediment due to the movement of tractors.
5. All the above contribute to a change in the fauna beneath or beside the farm.
6. Disturbance of the fauna (notably birds) of the area at low spring tides thus reducing the time available to them for feeding in winter.
7. Occupation of a valuable feeding ground for birds thus having an adverse effect on their survival and reproduction.
8. Occupation of a valuable roosting ground for birds.
9. Competition for phytoplankton with the indigenous inhabitants who are dependent on the same resource.
10. Adverse effects on several species of birds if the shore is cleared of rocks and seaweed.
11. Adverse effects on a number of species if large amounts of seaweed are collected for packing the shellfish.
12. There are serious ecological implications in the import of live Pacific oysters and their shell contents into the country.
13. It is possible that the presence of trestles will so alter sedimentation patterns as to change the entire nature of the ecotope.
14. Pacific oysters have spawned in Irish waters and could eventually establish a viable population. As they are generally a more tolerant species than the native oyster and they may well compete favourably thus displacing the native species.

1.6 Mitigating measures

1. The important and efficient way of ensuring that there is a minimum of conflict between ecological interests and shellfish farming interests is to choose a site wisely. This choice should be made on the basis of bird usage of the site, flushing of site, carrying capacity (phytoplankton concentrations and turnover).
2. Fallowing after harvest is one option to reduce the impact of organic enrichment and sedimentation on the substrate and the benthos.
3. Access points to the shore could be limited to restrict tractors driving on the beach. A minimal surface disruption route could be agreed at a local level between the farmers and the local ranger. Alternatively a tractor path to the trestles could be applied for in conjunction with the site licence application.

4. Original siting of the farm is critical as most adverse effects can be minimised or even prevented by making an informed decision at this stage.

5. One mitigating measure to counteract the ecological implications of the genetic transfer of Pacific oysters is to limit introductions to sterile individuals usually diploids and triploids. This also may have benefits from a husbandry point of view as sterile organisms tend to put energy into growth rather than reproduction (Shpigel, 1991; Guo *et al*., 1996). Triploids have advantages and disadvantages, they are a safe way to test *C. gigas* with little or no risk of reproduction. Use of F1 or greater progeny reduces the risk of disease. However they are genetically altered organisms and as such may pose risks to ecosystems receiving them. There is no clear consensus on whether field tests using triploids should be approved (Allen, 1993).

1.7 Research needed

There is a growing body of information on intertidal shellfish cultivation. Much is based on short-term studies or is speculative. Research needs will differ depending on the site but generally there are questions that need to be answered are related to birds and the benthos.

a. Birds

There is a lot of speculation, based on existing ecological knowledge of species, on the impact of intertidal shellfish cultivation on birds (O'Briain, 1993; Heffernan, 1995) without any field studies. Long-term studies would be very useful. These should include a baseline study to see which areas of the intertidal are being used by which species to feed, roost etc. before and after an intertidal oyster farm is introduced to see what changes in behaviour/site use occurs.

Investigations are also needed to see if a solitary tractor path would ease the effect of disturbance and whether intense disturbance in one area is preferable to a low level of dispersed disturbance.

b. Benthos

A long-term study would provide better information, on the affect of oyster culture on the benthos, than that which exists to date. It also would be preferable to have studies before and after the set up of a farm to document the changes and identify suitable biological criteria for use as environmental indicators in routine monitoring.

8. Management

Management strategies may be broken down into three sections:

a. Choice of site.

b. Management of individual farms.

c. Management of the ecotope (biological area).

a. Choice of Site

The choice of site is absolutely critical as this determines to a large degree the impact of the farm. The main factors from an ecological point of view would be

1. distance from feeding and roosting sites for birds.
2. current velocity and flushing times.
3. Phytoplankton concemtrations
4. Other demands for the same resources.

(See also Anon. (1991c) for guidelines on siting a Pacific oyster culture site)

b. Management of individual farms

In many respects this is the most important element. Once a farm is in place there will be various impacts associated with its mere presence. However, the impact that an individual farm will have on an area will depend to a large degree on the farmers' attitude. Site management practices cannot be undervalued, for example, some farmers welcome geese grazing on the *Enteromorpha* spp that grows on the oyster bags (P. Cullen, pers. comm.). On the other hand some farmers scare them away. Some leave equipment and abandoned trestles scattered on the shore (pers. obs.) and drive all over the shore. Other farmers have good sensitive and sensible housekeeping practices and respect all the other shore users. Education of, and interaction with, farmers should be a key management goal.

c. Management of the ecotope

Management of the entire ecotope (described above) e.g. a bay would ideally would fall to one or two authorities. This is not the case in Ireland. Management of a bay can fall under several different authorities with seemingly conflicting interests. The local authority has authority on land down to the High Water Mark, the Department of the Marine from below the High Water Mark. If the area is protected for wildlife the Wildlife Service is responsible for that and if the shellfish farms are grant aided by Bord Iascaigh Mhara they also have an interest in the site. When considering ecotope management all interests must be catered for. The critical point must be that no activity compromises another. This is the substance of sustainability.

There are strategies to help managers to cope with managing ecotopes. One of these is the single bay management (SBM) strategy which has been suggested for shellfish production areas (Bates, 1995). Another is ecosystem modelling. Raillard and Menesguen (1994) used this tool to estimate the carrying capacity of Marennes-Oleron Bay for production of *C. gigas*. Their model incorporated physical and biological processes; horizontal transport of suspended matter, feeding and growth of *C. gigas* and primary production. While these may help make decision making easier, it is critical to for managers to remember to take all interests into account.

MANILA CLAM (*TAPES SEMIDECUSSATUS*)

The only clam cultured in Ireland at present is the Manila clam (*Tapes semidecussatus* or *Ruditapes phillipinarum*); a non-native species. Current production is in the region of 100 tonnes (de G. Griffith, 1995). It can withstand temperatures ranging from 0 to 30 ûC, with optimal growth occurring between 18 and 25ûC (Britton, 1991). In Ireland, the average sea summer temperature is 13-16ûC and growth commences at 9ûC provided food is not limiting. The Manila clam is tolerant of a wide range of salinities with the optimum lying between 20 and 34 ppt. It requires more wave-sheltered conditions than mussels and oysters (Britton, 1991).

1.1 Method of cultivation

The site should have a short fetch in the path of the prevailing wind direction. Suitable areas for cultivation are frequently inner bays and estuarine areas. Middle estuaries are commonly the most suitable as the substrate usually consists of firm or silty sand, and the salinity of the water tends to be higher than in the inner estuaries (Britton, 1991). In order to maximise feeding and growth the clams need to be covered by seawater for a maximum period. Therefore, the area of interest for farming lies between mean low water neap tide and mean low water spring tide level (Britton, 1991). Clams need to be buried in sand in order to develop a normal shell, and harvesting (by ploughing) would prove practically impossible in any other intertidal substratum (C.Duggan, pers. comm.).

As the Manila clam is a non-native species all seed must be supplied by hatcheries. The main supplier in Ireland is Lissadell Shellfish Company, Co. Sligo. The hatchery seed is grown off-bottom between mesh in flat wooden frames. These are raised on wheels and can be manoeuvred up and down the shore like a wheelbarrow. The seed are cultured in this way until they reach a size of 8-10 mm, which usually takes one year. Once they reach this size the method of on-growing is known as the plot system. This involves rotovating the site to remove predators (e.g. crabs), ploughing the sand and sowing the juvenile clams. The clams are then protected from predators by incorporating a strip of netting over them (Britton, 1991). The largest clam growing operation in Ireland is based at Lissadell, Drumcliff Bay in Co. Sligo.

Clams are usually grown further up the shore than oysters and they are commonly grown on the same shore. This arrangement is possible as the oysters are grown about 0.5 m off the ground on trestles, whereas the clams are dug into the sand. The width of the area available for shellfish farming will be dictated by the gradient of the shore and the tidal amplitude (Britton, 1991).

1.2 Data available

Practically everything published on the ecological impacts of cultivating Manila clams has been speculative until this year. The preliminary results of a field experiment carried out by researchers from the Directorate of Fisheries Research (MAFF) were published earlier this year. There is much research needed on this topic as some believe that there are substantial ecological impacts associated with this cultivation (O'Briain, 1993). However, proper studies are needed to calm, or reaffirm, these concerns.

1.3 Ecological impacts of cultivation of *Tapes semidecussatus*

From the evidence available, and unlike many of the other species dealt with to date, the impact of Manila clam cultivation has little to do with the species being cultivated and much to do with the method of cultivation.
The ecological impacts of cultivating and harvesting the Manila clam will be dealt with under three headings: local impacts, regional impacts and harvesting impacts.

LOCAL IMPACTS

Local impacts will be discussed under the following headings:
1.3.1 Water composition.
1.3.2 Biodeposition and sedimentation.
1.3.3 Effect on benthos.
1.3.4 Impacts of site preparation on the foreshore.

1.3.1 Water composition

There is no data on how the cultivation of Manila clams impacts on water composition. However, ammonia excretion is expected, as all bivalves dealt with in this review excrete ammonia into the water column.

1.3.2 Biodeposition and sedimentation

None of the papers consulted referred to biodeposition by Manila clam. The only organic enrichment noted was associated with increased sedimentation due to the anti-predator netting (Spencer, 1996).

1.3.3 Effect on benthos

The use of protective plastic netting increases sedimentation rate and leads to an increase in the abundance of deposit feeding infauna (Kaiser *et al*., 1996; Spencer *et al*., 1996 b). Kaiser *et al.* (1996) results from a site of commercial clam cultivation in Kent, England, show that diversity was not affected by clam cultivation. However, while this was the case for this site it may not prove to be the case for any other sites. The control areas were dominated by the polychaete *Nephtys hombergii*, whereas the clam lay was dominated by deposit feeding worms *Lanice conchilega, Euclymene lumbricoides* and the bivalve *Mysella bidentata* (Kaiser *et al*., 1996). This agrees well with the Exe estuary site where a similar distribution of species was found, although the species differed i.e. deposit feeding worms (*Ampharete acutifrons* and *Pygospio elegans*) and bivalves dominated the net covered plots. Yet again unnetted control areas were dominated by *Nephtys hombergii* (Spencer *et al*., 1996a). Neither study investigated the effect of intertidal clam culture without nets due to legal constraints governing introduced species.

Simenstad and Fresh (1995) noted that in general most epibenthic crustaceans were depressed under the predator exclusion nets compared to the unnetted control. Densities of adult harpactoid copepods were consistently higher (though not statistically significant) in the control plots. In contrast, they often found enhanced epibenthos densities on their netted plot at their other sites (Simenstad and Fresh, 1995). So the conclusion must be that impacts of clam culture/netting may be largely site specific.

Spencer *et al*. (1996) looked at the ecological effects of intertidal Manila clam cultivation at the end of the cultivation phase i.e. 2.5 years after original planting. Over the full 2.5 years the presence of the netting was shown to increase sedimentation which elevated the ground profile by about 10 cm and caused a small but significant increase in the percentage of fines and organic content of the sediment. The netting also encouraged higher densities of some species of infaunal deposit feeding worms which became the dominant fauna. During the first six months of cultivation the fauna was dominated by the opportunistic spionid, *Pygospio elegans*. After one year, the stabilising effect of the netting on the sediment led to the establishment of species such as *Ampharete acutifrons* and *Tubificoides benedii,* which displaced *P. elegans* as the community dominant. The observed biological responses indicated that organic enrichment occurred within net covered areas. The magnitude of community change is far less than that which occurs in association with some other marine culture practices which create anoxic sediments and impoverished infaunal communities. However the results may underestimate the impacts given the fact that out of an original $500m^{-2}$ of Manila Clams planted only $26m^{-2}$ survived.

In Washington, USA, the method of clam cultivation is similar to Ireland but they add gravel to mudflats and sandflats to enhance clam production (Simenstad and Fresh, 1995). This may subtly impact certain benthic and epibenthic invertebrates without changing the carrying capacity for estuarine dependent taxa according to Simenstad and Fresh (1995). Thompson (1995) and Thom *et al*. (in press cited in Simenstad and Fresh (1995) indicate that substrate modification for enhanced clam production can significantly depress cover of macroalgae, enhance Chlorophyll *a* concentrations, increase benthic respiration and increase nutrient fluxes (particularly PO_{3-} and total inorganic nitrogen and , NO_2 and NH_{4-}). The magnitude of these responses tend to be very site specific .

1.3.4 Impacts of site preparation on foreshore species

Preparation of the shore for intertidal clam cultivation, involves clearance of seaweed, rocks and stones (O'Toole, 1990; O'Briain, 1993). Seaweed and rocks represent a major floral and a major faunal habitat (Barrett and Yonge, 1984). Other methods of preparation for cultivation are of concern, especially methods employed for clam cultivation. The French farmers cover plots with black plastic to kill any fauna which might compete with, or prey on, the juvenile clams (Dravers, 1988). In Ireland, the site is usually rotovated before laying clams to remove predators, such as crabs, and the sediment is ploughed to facilitate sowing of the juvenile clams. Another practice is the removal of local mussel beds to prevent spat settlement on oyster bags or nets (O'Briain, 1993). Mussel spat settling on cages reduces food supply by blocking water inflow, and clumps the cultured shellfish together depriving the centre ones of food. It also makes the clams difficult to separate and consequently difficult to market.

The removal of stones and seaweed from the intertidal zone effectively limits feeding habitat for such species as turnstones (O'Briain, 1993). This practice also affects, albeit to a lesser extent, brent geese, wigeon, teal and pintail. The Turnstone is particularly sensitive to rock and seaweed removal by shellfish farming as it feeds on the organisms which live on and under seaweed and stones. The turnstone and the curlew may also suffer due to crab removal by clam farmers. Crabs make up a substantial proportion of these birds' diet (Heffernan, 1995).

Removal of local mussel beds will also affect many species of birds. The most adversely affected are likely to be oystercatchers which depend on bivalves as a substantial part of their diet. Removal of these beds will also affect Knot as this species feeds on mussel spat (Cramp and Simmons, 1977). Rocks provide a suitable substratum for mussels to attach to and crabs to shelter beneath. Rock removal could affect oystercatchers, curlews and turnstones who feed on these species.

REGIONAL IMPACTS

The regional impacts refer to those impacts which affect a larger area that the immediate farm area. These will be discussed under the following headings
1.3.5 Phytoplankton depletion
1.3.6 Transfers of organisms

1.3.5 Phytoplankton depletion

Although there is no information regarding phytoplankton depletion by Manila clam it is assumed that the impacts will be similar to those recorded for other filter feeding bivalves e.g. mussels.

1.3.6 Transfers of organisms

In the long-term it is possible that Manila clam spawning could lead to the production of a self-sustaining population may occur due to;
1. Acclimatisation.
2. Elevated temperatures in sea lochs.
3. Movements in water currents and isotherm patterns (Meikle and Spencer, 1989).

Concerns have been expressed about the likelihood of escapes from intertidal Manila Clam culture plots where only a top containment net is used (Meikle and Spencer, 1989). The Manila clam was recorded as spawning in Sligo in 1989 (Burnell and Cross, 1989).

1.3.6 Impacts of cultivation on foreshore species

Probably the impact of greatest concern regarding clam cultivation is habitat loss. Habitat loss is the best known mechanism for species extinction. In this case the concern principally applies to bird populations as areas which would normally be available for feeding and roosting may be occupied by clam culture. Loss of habitat can arise from the presence of structures used for growing shellfish on intertidal feeding grounds. These structures include frames used for holding small spat, and areas under netting. The farming operations are generally quite small in terms of area covered (1-2 ha.). However, the cumulative reduction of feeding grounds arising from the increasing number of operations can be substantial (O'Briain, 1993).

As clam cultivation in Ireland is confined to sandy estuaries, birds feeding on sand would be most likely to suffer from loss of habitat. Species most likely to be affected by clam farming are oystercatcher, sanderling and bar-tailed godwit as all have preference for feeding on sandy habitats. Species which may lose roosting habitat only, are the golden plover and some species of geese (Heffernan, 1995).

The presence of the netting on the foreshore has its own effects not only are birds denied access to their habitat of choice but the same is true for crabs and fish although there appears to be no published information on this. The netting acts as a substrate for the growth of green seaweed. This attracts a higher abundance, than non-netted areas, of periwinkles which graze on it (Spencer, 1996). The netting, rather than the clams, also encouraged a higher abundance of some sedentary worms but without affecting the overall level of species diversity. This may be a positive benefit to animals higher up the food chain e.g. crabs and birds (Spencer, 1996). The organic content of the sediments beneath the nets is higher, either with or without clams. This is probably due to the natural decay of *Enteromorpha* fouling the nets or from the faces of the periwinkles grazing on the weed (Spencer *et al*., 1996a).

Clam cultivation requires tractors for maintenance and to carry bags, boxes and trestles between different areas of the shore. This may result in tractor tracks over a wide area of shore. This can lead to compression and churning up of sediments in the intertidal zone with negative effects on the invertebrate fauna (O'Briain, 1993). If plants such as *Zostera* , are present on the foreshore, they could also be damaged by tractor activities, with concomitant effects on birds such as brent goose and wigeon that feed on this plant (Cramp and Simmons, 1977).

Disturbance from intertidal shellfish farming is mainly caused by the presence of tractors and groups of people working on the mudflats (O' Briain, 1993). Activities on the mudflats include brushing of weed off clam nets and harvesting (pers. obs.). For more details see the section on disturbance under Pacific oysters.

Once the shellfish are harvested they are often packed in seaweed (BIM, 1983). Depending on tonnage produced, a significant amount of seaweed may be removed from an area (see section on seaweed harvesting page 102). This practice will affect birds and other animals dependant on this habitat.

1.4 Harvesting

Harvesting in Ireland is carried out by a modified potato harvester (pers. obs.). There is no data in the literature on the impacts of such a method. However, there is some data on the impacts of hand-raking versus suction dredging. The current method most probably lies somewhere between these two extremes.

a. Harvesting and the sediment

The expected impact would be similar to dredging (see page 59) i.e. that finer sediments get washed away in the sediment plume thus reducing productivity. Harvesting may lead to adversly altered habitats and reduced juvenile recruitment but the impact will depend on the type of dredge used. Harvesting can be of benefit to predators as it leaves fauna open to predation as their environment is drastically changed.

b. Harvesting and the benthos

As the harvesting mechanism gathers the target species it would be expected that a proportion of these and indeed many of their associated organisms would be damaged or killed by the operation. Spencer (1996) observed that harvesting by handraking caused about 50% reduction whereas suction dredging caused about 90% reduction in species diversity and abundance. The former took 3-4 months, and the latter 8 months, for site recovery. This study however only looked at a site after its first clam harvesting event and does nothing to address the question of the impacts of large areas being harvesting every 2-3 years long-term (Spencer, 1996).

Intertidal cultivation of clams also interferes with the habitat in the same manner through ploughing of the intertidal zone. Ploughing the sediment for sowing and harvesting is also likely to have a detrimental effect on invertebrates present. In 1988 in Drumcliff Bay, Co. Sligo, 2.5 million clams (each weighing 8-12 g) were harvested from approximately 1.5 hectares of beach by ploughing (Anon., 1988). Such concentrated activity every three years is likely to have significant localised effects.

One study on the ecological consequences of mechanical clam harvesting found that although clam harvest did not affect either the density or species composition of small benthic macroinvertebrates, *Zostera* beds were badly affected. In mechanically harvested areas, four years later, *Zostera* was 35% lower than controls (Peterson *et al*., 1987). This study related to the method of "clam-kicking" which involved using the backwash off a motor attached to a boat in shallow water to suspend the clams (Peterson, 1987). This method is not practised in Ireland but the intertidal zone is ploughed, and this severely disrupts the sediment. Ploughing the sediment in areas where there are *Zostera* beds is likely to cause serious damage to this plant (Simenstad and Fresh, 1995). This may have long-term effects on numbers of brent geese and wigeon.

Kaiser *et al*. (1996) noted that although harvesting removed a large proportion of the infauna, seven months later it was not possible to differentiate between the harvested and control areas. They concluded that clam cultivation increases productivity within limited areas during the growing process and the effects of harvesting do not persist for more than seven months (Kaiser *et al*., 1996). However, an increase in productivity is not necessarily positive and in an aquatic environment is almost always undesirable. Mattsson and Linden (1983) observed that a rich community of starfish and brittlestars were replaced by opportunistic polychaetes underneath a suspended mussel culture farm. The biomass may have increased but the conservation interest and value certainly decreased. Futhermore, as the experimental site (for both the test and the control) is a site of commercial clam cultivation and the control may also have been subject to some local impacts of clam cultivation. In addition, there may be knock on effects, for example, as recruitment and recolonisation may be dependent on source areas such as this control site.

1.5 Hydrology

Netting is likely to reduce the water flow at its seabed interface causing an increase in the sedimentation rate of organic and fine material (Spencer *et al.*, 1996).

1.6 Summary

1. The anti-predator netting covering the plots of Manila clams are responsible for most of the impacts on the benthos as they increase sedimentation and % organic material.
2. The netting resulted in an increase in the numbers of deposit feeding worms.
3. Removal of rocks, seaweed, crabs and local mussel beds can have serious knock-on effects on the ecology.
4. Phytoplankton depletion is likely given sufficient numbers of clams.
5. There is a small, but unlikely, possibility that the Manila clam could become established in Ireland.
6. Loss of habitat to birds feeding and roosting on sand is a high probability.
7. Tractors on the shore can result in the churning up of sediments with losses in flora and fauna.
8. Harvesting is likely to lead to losses of non-target species.

1.7 Research needed

The research that is needed is similar to that needed for Pacific oyster cultivation (see page 68).

1.8 Management

The management that is needed is also similar to that needed for Pacific oyster cultivation. Spencer (1996) recommends the following mitigating measures to reduce the environmental effects of clam cultivation:

1. Ensure that netting is kept in good repair to prevent the escape of these non-native species.
2. Where possible harvest by hand raking to reduce disturbance of the sediment.
3. Where possible, harvest between autumn and spring to ensure that natural invertebrate settlement can quickly regenerate communities affected by harvesting. In contrast, this is the worst time to harvest as overwintering birds will be disturbed or a proportion of their prey destroyed in harvest. Probably either extreme i.e. early autumn or late spring would be optimal given all considerations.

These measures do not really address the potential problems caused by clam farming especially as they do not consider the impact on birds. Again, the mitigating measures for Pacific oyster cultivation also apply to clam cultivation.

OTHER SHELLFISH/CRUSTACEANS CULTURED IN IRELAND

Lobster (*Homarus gammarus*)

Lobster culture is currently underway in Ireland. The system is more a lobster enhancement project than pure culturing. The waters are stocked with lobsters from a hatchery and the berried (egg bearing) females are V-notched. i.e. a small section is removed from their just below their tail fan making them recognisable even after a large number of moults. This is one way of protecting the breeding stock. The co-op in Wexford released two batches of 4000 juveniles (12-13 mm in length) in 1995. As well as the juveniles, the co-op has released a couple of thousand lobsters marked with a V-notch. The Irish government has introduced legislation which makes it an offence for any person to retain or have in their possession a lobster marked with a V-notch in the tail or mutilated (Edwards, 1995). The ecological impact will depend on the resources being contested by these extra lobsters.

Abalone (*Haliotis tuberculata*/*Haliotis discus hannai*)

Abalone are not native to Ireland. There are two species of this marine gastropod which are of interest to Irish aquaculture, these are the European abalone (*Haliotis tuberculata*) and the Japanese abalone (*Haliotis discus hannai*). Only the former is at present grown commercially in Ireland (Hensey, 1990), though trials have shown the latter to adapt well to Irish waters and to grow 10% faster than the European abalone La Touche *et al*., 1993).

One of the reasons for the late development of the abalone industry in Ireland is due to its non-native status in Ireland. It inhabits the west coast of Africa to the English channel island. After eight years quarantine abalone was finally considered safe to introduce to Ireland in the late 1970's. Production is limited by spat availability and production in 1994 was only a few hundred kilos (Charron, 1995). European abalone has been grown in small quantities since the 1980's is fed a mixture of dillisk and kelp (Bates, 1993).

Abalone are macrophagous, i.e. they are grazers which feed on seaweed. They occur naturally along exposed coastlines and show preference for areas with clear oceanic water. Clear water is essential as their elaborate gills are unsuited to turbid waters where high silt loads could clog their gills. Abalone in culture require either a very sheltered environment or cultivation on sub-surface longlines as they show poor growth rates with excessive disturbance of their cages. Proper exchange of water is essential to remove waste products, as build up of waste products can also lead to reduced growth. They are fully marine animals and as such inhabit waters with near to full salinity with a minimum of 30 ppt. The temperature range for the European abalone (*Haliotis tuberculata*) is 7-18ûC with an optimum at 15-18ûC; the Japanese abalone (*Haliotis discus hannai*) has a range of 6-25ûC with an optimum at 15-20ûC (La Touche *et al*,. 1993).

As it is a non-native species all the spat must be supplied by hatcheries. There are currently three active abalone hatcheries in Ireland. The spat are grown in tanks and fed on algae (Hensey, 1995). To date in Ireland spat has been sold only in the size range 10-15 mm (La Touche *et al*,. 1993). At this stage they are ready to be transferred into sea-cages, these cages are made from modified 250 l barrels divided into three compartments and with three equally spaced windows with a changeable mesh. The barrels are suspended from longlines, rafts or submerged longlines. The juveniles are fed seaweed through a porthole and on reaching 26 mm they are ready for on-growing (La Touche *et al.,* 1993). There is only one method, used in Ireland (Clew bay, Co. Mayo) for abalone on-growing and that is bottom laid cages. The cages are made from modified lobster pots and are laid in strings on the sea floor in suitable areas and are hauled to the surface for feeding and cleaning. Each string consists of 8-10 pots connected to each other by 4 metre lengths of leaded rope. The cages are stocked with abalone of mean size not less than 30 mm as survival of smaller sized ones is lower (La Touche *et al*., 1993).

Throughout the farming period abalone must be fed with seaweed. Trials have shown that the red seaweed *Palmaria palmata* gives the best growth performance, a kelp *Laminaria digitata* produced inferior growth but has the advantage of sheer abundance (La Touche *et al*,. 1993). About 20 kg of (wet weight) of seaweed is required to produce 1 kg of abalone (Hensey, 1995). Viana *et al*., 1996 looked at the possibility of using silage made from fish and abalone viscera as an ingredient of abalone feed. This proved to increase growth rates compared to a kelp based diet. However, the dangers of feeding animals their own offal is well documented for example in the transmission of BSE (Bovine spongiform encephalopathy) in cattle.

The main ecological implications of growing abalone are likely to be the following

1. Risk of establishment of a non-native species in Ireland, thus occupying an ecological niche and competing for already limited resources.
2. Increased pressure on seaweed resources which may not cause any problems at present, but may do so in the future.

Sea Urchin (*Paracentrotus lividus/ Psammechinus miliaris*)

There are two native species present in Ireland, the purple sea urchin (*Paracentrotus lividus*) and the green sea urchin (*Psammechinus miliaris*) (Moyihan *et al*., 1991). Sea urchins are macrophagous, i.e. they graze on algae with a preference for kelp. They are generally considered to be herbivores but studies have shown that food of animal origin is accepted. They can also absorb dissolved substances across their external membranes e.g. amino acids (LeGall, 1991). The green sea urchin is able to tolerate more freshwater than the purple sea urchin. The purple sea urchin (*Paracentrotus lividus*) lives here at the northern limit of it's range which seems to be determined by a sea temperature of 8ûC in February (LeGall, 1991). For *Psammechinus miliaris* death occurs rapidly at temperatures below 1ûC and above 22ûC, the optimum temperature is 13-17ûC. A similar pattern exists for the purple sea urchin (*Paracentrotus lividus*) although the optimum temperatures are slightly higher (LeGall, 1991). Sea-urchins need a hard solid surface, usually a rock crevice or holes, which they bore themselves, especially in limestone. Urchins placed on substrates to which they cannot cling die fairly rapidly (LeGall, 1991). In nature they occupy sublittoral habitats from permanent rock pools, at the low tide levels, to boulder strewn sea-bed at 30 m (Moyihan *et al*., 1991).

In Ireland there has been a lot of research into the feasibility of culturing purple sea urchins (*Paracentrotus lividus)* on a commercial scale, but none is published to date. If commercial cultivation was to be carried out here it would be carried out using the same/similar methods to those for abalone aquaculture (Moyihan *et al*., 1991).

The main ecological concern would be the harvesting of kelp. As they are an overfished native species ranching could be considered a form of restocking. This is provided the spat is sourced in Ireland from native wild stocks.

SECTION 4

INTERTIDAL HARVESTING

INTERTIDAL HARVESTING

In Ireland, intertidal harvesting is carried out by hand. The main organisms harvested are cockles, sea urchins, mussels, lugworms, periwinkles and seaweed. None of these activities require licences, yet they can have quite a dramatic effect on the ecology of the areas affected. The effect will depend on the method and intensity of harvesting.

Lugworms/Ragworms (*Arenicola marina*/*Nereis virens*)

The ragworm *Nereis Virens* and Lugworm *Nereis Virens are* polychaete worms of commercial importance, through their demand as bait. Van den Heilegenberg (1987) estimated that 323 million lugworms are collected annually in the Dutch Wadden Sea. James *et al*. (1993) estimated that about 8 million are collected each year from the north Norfolk coast. Commercial bait digging may lead to severe disturbance of natural environments, to conflicts of interest with other resource users and with the needs of conservation (Olive, 1991).

Hand Harvesting

Professional bait diggers dig 50m^2/tide in the Dutch Wadden Sea (van den Heilegenberg, 1987). In Norfolk it has been calculated that diggers turn over a 50 m strip 6 km in length each year (James *et al*., 1990). At times of peak demand for fishing bait more than 100 collectors have been recorded digging on a 6 km stretch of Norfolk beach (James *et al*., 1993).

Digging for lugworm for use as angling bait has several adverse environmental effects. The sand is overturned and the natural communities disrupted (Hartnoll, 1990). James *et al*. (1993) looked at the influence of bait digging on intertidal marine benthic invertebrates on two eastern English estuaries. They noted that the densities many non-target organisms such as *Nephtys caeca*, *Lanice conchilega*, *Cerastoderma edule* and *Nematoda* were signifcantly reduced by digging (James *et al*., 1993). *Macoma balthica, Ostracoda*, *Harpacticoida* and *Foraminifera* densities were not significantly affected.

Cox (1991) discovered that a negative impact associated with bait digging was the depletion of larger infaunal species, especially those sensitive to disturbance, in previously undisturbed sediments. James *et al*. (1993) discovered that bivalve survival was related to mobility within the substratum. Field experiments have shown bait digging to produce high mortality in cockles (Jackson and James, 1979 cited in Hartnoll, 1990), and a general reduction in species which may serve as fish food (Van den Heiligenberg, 1987).

McLusky *et al*. (1983, cited in Van den Heiligenberg, 1987) found a reduction after hand-digging of 80-100% in the mud snail *Hydrobia ulvae* and near 100% in the bivalve *Macoma balthica*. These are very high reductions and are attributed to the diggers leaving the sediment in a mound and not filling the basins again. *Hydrobia* is the principle food source of shelduck and care must be taken not to destroy this resource.

Cox (1991) noted that bait digging caused disruption of sediment layers with anaerobic sediments being re-distributed to the surface, so smothering the infauna. He also noted that this lead to an increase in the abundance of small infaunal species tolerant to anoxic conditions. This disruption of the sediment also results in heavy metals bound in the anoxic layers being made bio-available.

James *et al*. (1993) looked at the effect on the substratum and found no differences in particle size distributions of the substrata in dug and control plots. They noticed that the mounds and hollows produced in marine sediments by digging may disappear quickly or remain for months. They concluded that the process creates a different textural environment, with coarse-grained sand mounds and basins surrounded by finer, silt-rich sand.

Mechanical Harvesting

Beukema (1995) carried out a study over several years to look at the long-term effects of mechanical lugworm harvesting. He reported that within an area of 1 km^2 on Balgzand in the Netherlands, the lugworm mortality rate practically doubled, due to dredging over a four year period. This resulted in the population being reduced from twice the overall Balgzand average to a value close to the average. Simultaneously, total zoobenthic biomass declined even more by the complete extinction of the population of clams *Mya arenaria* that initially comprised half the total biomass. Of the other, mostly short lived species, only *Heteromastus filiformis* showed a clear reduction during the dredging period. Recovery of the biomass of the benthos took several years, particularly by the slow re-establishment of a *Mya* population with a normal size and age structure. Beukema (1995) reported that although the structure of the benthic community remained incomplete for several years, the functioning of the community appeared to be hardly affected. No major functional group had disappeared and animal production remained close to its original level.

Van den Heiligenberg (1987) compared mechanical and hand-digging methods of bait collection. He showed that digging for lugworms, mechanically or manually, has a severe impact on other macrobenthic organisms. However, hand diggers caused far less mortality (1.9 g), compared to mechanical harvesters (up to 13.4 g), of other benthic organisms per gram of lugworm harvested. After six months or less most species had completely recovered. All small sized species (the polychaetes *Scoloplos* and *Heteromastus* and the bivalves *Cerastoderma* and *Macoma*) had recovered, but not lugworms populations or total biomass. Some of these non-migrating animals such as *Heteromastus filiformis* died from asphyxiation or desiccation. Recovery is dependent on two processes; recruitment of juveniles and migration from adjacent unaffected areas (Van den Heiligenberg, 1987).

James *et al*. (1993)noted that bait digging distorts species composition and reduces biomass of the infauna. These changes impact on species outside the infaunal community, especially epibenthic predators such as birds and fish. According to Van den Heiligenberg (1987) digging areas close to the shore will become more or less permanently unsuitable for some bird species due to disturbance and damage to the sediment fauna.

The presence of people digging in the intertidal zone at low tide is also likely to seriously disturb feeding wildfowl in winter. In Lindisfarne, National Nature Reserve and Special Protection Area, in the 1980's large numbers of bait diggers in the wildfowl refuge area are thought to be responsible for the greatly reduced use of the area by several species of wildfowl (Townshend and O'Connor, 1993). In contrast, Burger (1981) noted that the presence of clammers in Jamaica Bay, New York, did not usually cause the birds to flush. In Baldoyle Special Protection Area for birds in Co. Dublin Ireland, lugworms are collected in the winter and ragworms in the summer. In order to protect the birds from the disturbance, to some degree, an area has been zoned for this activity (L. Patton, pers. comm.).

SUMMARY

Harvesting of lug/ragworms has the following effects:

1. A severe impact on the benthos whether harvesting is by hand or by machine.
2. By hand is by far less harmful to the non-target sediment fauna.
3. The species affected in the short term (< 6 months) are *Hydrobia*, *Scoloplos*, *Heteromastus*, *Cerastoderma* and *Macoma.*
4. Harvesting may result in the sediment becoming coarser in the long term.
5. Harvesting may destroy/damage food sources for both birds and fish.
6. Manual or mechanical digging will cause disturbance to wintering wildfowl feeding.
7. Harvesting may causes an impact which can take years to recover for the large species.

MANAGEMENT PRESCRIPTIONS

1. If there is a choice of harvest method hand digging is preferable in relation to damage to benthic fauna.
2. Diggers should be required to refill dug basin to protect the infauna.
3. Consideration must be given to disturbance caused to birds by bait digging in an area of conservation. Areas may have to be sacrificed to concentrate the activity rather than having a low level of disturbance throughout the site.

Cockles (*Cerastoderma edule*)

In Ireland at present cockles are collected by hand raking. Most of the data available relates to mechanised methods and although this is not relevant at present it is likely to be in the next decade and so is included in this discussion. Hand-raking may expose some of the smaller cockles on the surface with a greater risk of depredation, but probably causes little extra mortality in cockles or other species (Hartnoll, 1990). This observation was also made in relation to tractor dredging of cockles in the Burry inlet (Cotter *et al*., 1993). Although some birds may benefit in the short term, cockles form a major component of the diet of oystercatchers and the extensive fishing of cockle beds will reduce the available food supplies (Hartnoll, 1990). This has been borne out by studies in the Wash Estuary, which have shown that the density of oystercatchers fell with reduced availability of cockles (Goss-Custard *et al*., 1977 cited in Hartnoll, 1990).

Another method of harvesting cockles is by suction dredging. This method is practised in England and Scotland. The suction dredger works by removing the top layer of sand and filtering it to retain the larger items. This is likely to disrupt communities, damaging sediment fauna other than the target species and causing above-average mortalities (Hartnoll, 1990). In contrast, using a tractor dredger Cotter *et al.* (1993) were unable to show a significant difference between mortalities in their dredged and control sites 14 and 92 days after dredging.

Rosten (1995) looked at the effects of mechanised cockle harvesting on the invertebrate fauna of the flats of Llanrhidian Sands, Wales, UK. This area has been harvested by hand raking since mediaeval times and this experiment was designed to assess the likely impacts of licensing such a method. Two very different sites were chosen.

Site A consisted of poorly sorted sand within the range 2.5-3.0 diameter. The surface was uneven, with small pools, casts of the lugworm *Arenicola marina*, small burrows, and the trails of the small snail *Hydrobia ulvae*. Brown diatom/algal growth was present on the surface.

Dregding site A adversely affected *Pygospio elegans,* which had not recovered to pre-dredge levels even six months after the dredge event. The polychaete *Scoloplos armiger* disappeared completely from some of the dredged plots but not the adjacent control plots, although six months later the population had stabilised. The distribution of the polychaete *Nephtys hombergii* was disturbed by dredging but quickly recovered. *Hydrobia ulvae* suffered a large decline and recovery was remarkably slow for an allegedly mobile animal. However, there was no statistically significant difference between dredged and control plots six months later. The cockle populations, as expected, were significantly reduced and continued to be reduced right until the end of the experiment.

Site B was situated within an area of well sorted, fairly coarse sand of 2.0-3.0 diameter. It was well drained and analysis showed that sediment parameters were not altered by dredging. Results showed that natural fluctautions were greater that those resulting from dredging. Small interstitial forms showed no decline and in some cases (*Hydrobia Ulvae, Cerastodermata edule* and *Pygospio elegans*) decline in experimental plots was mirrored by a more gradual decline in control plots. This difference between sites appears due to site conditions.
The reasons suggested for the delay in the recovery at site A are that harvesting destroyed the stable diatom covered surface (which is food source for many organisms), permanent tube dwellings, eggs on in or attached to the sediment and altered the sediment type. These factors apply to all disturbances to the sediment be they bait digging, dredging or other forms of harvesting. The fauna at site B appears to be more dynamic, and perhaps better able to adapt to dynamic sediment conditions.

SUMMARY

1. There are three methods of harvesting cockles: by hand, by tractor and by suction dredger.
2. All have some impact on the non-target infauna. The impact will, however, depend on the scale and intensity of the harvest and well as the sediment type at the site.

MANAGEMENT

The management prescriptions are similar to those for lugworm harvesting (Page 97).

Seaweed Harvesting

The Irish seaweed industry employs nearly 500 people, exports 85-90% of its produce and had a turnover of 3.5 million in 1995 (Guiry, 1996). In Ireland, we have over 500 species of seaweed, but most of the biomass is provided by a relatively small number of species: the kelps (*Laminaria* and its relatives), wracks (*Fucus* and its relatives) and the detached coralline algae known as maerl. Wracks are largely found in the intertidal whereas kelps are confined largely to the subtidal(Guiry, 1996). This review will deal with the kelps, the wracks and the red algae separately as they are harvested in different ways. Maerl will not be dealt with as it is not collected intertidally. It must be remembered that any subtidal extraction of maerl, and of other seaweed, will have an impact on the intertidal zone.

At present all seaweed harvesting is by hand. However, with increasing interest in these resources, this is likely to change in the near future as these harvesting technologies currently exist in other countries. At present the ecological impact of harvesting these species is likely to be very small as the resource is under-utilised. However, with mechanisation the risks of ecological impact will increase substantially. This review aims to look at the species currently harvested and (where the information exists) present information on the current ecological impact of harvesting. Furthermore, where applicable, present data from other countries on methods of harvesting and impacts observed is outlined. Where management strategies have been noted or mitigating measure discovered these will also be presented.

ASCOPHYLLUM NODOSUM

In Ireland, exploitation of seaweed is almost entirely centred on *A. nodosum*. The main areas of collection are: Connemara (Co. Galway), Co. Mayo, Co. Donegal and the Shannon estuary (Guiry and Blunden, 1981). In these areas, the topography does not always make it possible to reach the harvesting sites directly, and small boats are often used for access. Harvesting is by hand, the algae is cut with a sickle or a knife just above the holdfasts. The bladders of *Ascophyllum Nodosum* allow the cut seaweed to float and it may be enclosed by nets or ropes. Each net can hold about one tonne of seaweed, and the daily output is 7-8 tonnes per person (Briand, 1991).

Cullinane (1984) surveyed the accessible intertidal fucoids along the west coast of Ireland from Cork to Donegal, a distance of 1,950 km. He estimated the dry weight of accessible *Ascophyllum* to be 41, 901 +/- 11 563 tonnes . He concluded that given the low prices and a lack of harvesting tradition much of the weed will remain unharvested. However, a more detailed recent study confirmed that in excess of 130,000 harvestable tonnes of *Ascophyllum* are available on the west coast i.e. the

Galway-Mayo region and in Donegal, of which 34,000 tonnes were taken annually (Guiry, 1996). This is a source of seaweed meal and is also exported to Scotland for alginate production (Guiry, 1996).

Many researchers have noted that *Ascophyllum* can be sustainably harvested by leaving a stump of the plant behind to regenerate after harvesting (Baardseth 1955, 1970; Keser *et al*., 1981; Guiry 1996). Baardseth (1970) noted that in Galway cutting of *Ascophyllum* took place in Spring and the cutter left parts of the plants behind. Such a cut (about 10 cm from the ground) or so resulted in the establishment of a number of stumps which could produce new shoots. He noted that the annual harvesting of this species in Ireland is centuries old without degeneration of the resource being noticed. Keser *et al*. (1981) experimentally cut *A. nodosum* plants at holdfast level, 15 cm and 25 cm in length. Their results show that regrowth of *A. nodosum* after three years following cutting to the 15 and 25 cm level was over 95%. This resulted from the greater number of potential regeneration points created by the younger, highly branched plants. Currently in Ireland *Ascophyllum* is generally cut on a 4 year cycle with enough material (usually a stump 4-6 cm long) left behind after cutting to assist in regrowth (Guiry, 1996).

Baardseth (1955) believes that this constant production from the base is far more important in maintaining the population of *Ascophyllum* than the regrowth from fertilised eggs. The reason is that single shoots from an egg have found to be rare, despite the huge quantities of eggs released each spring. New lateral or basal shoots will be initiated from cut branches and growth will continue at these new growing tips. Growth will also continue at all uncut branches. If a clump of laterals is cut to less than 10 cm from the holdfast, then very little lateral growth will occur on the truncated shoots (Baardseth, 1970).

Another issue relating to sustainable harvesting is the length of the harvesting cycle i.e. how many years does it take for the resource to regenerate. At present the Irish stock is cut on a four year cycle. This is just within the range recommended by Baardseth (1955). He reported that five or six years between each cutting would seem most profitable, although he conceded 4/5 years as probably sufficient given the conservative nature of his estimates. His calculations were for Galway Bay and the optimal time period between cuttings is likely to change with location.

The rate of recovery will vary from site to site and will depend on the degree and intensity of harvest as well as the environmental conditions. Based on simple productivity measures under a variety of environmental conditions and depending on the initial biomass, recovery time can be 1-5 years. Keser *et al*., (1981) also looked at the affect of annual harvesting, in Maine USA, and noted that the biomass harvested declined with successive harvests. Staggering and annually alternating the harvest area provides one means of minimising this problem. Baardseth (1970) noted that in Europe, harvested stands require 5 years to recover following exploitation rates of 98%, and 3 to 4 years after an exploitation rate of 70-80%. Sustainable harvesting is inextricably linked to the annual production by the crop and methods of estimating this were investigated by Cousens (1984) who discovered that production of *Ascophyllum nodosum* ranged from 0.61 to 2.82 kg dw m^{-2} in Nova Scotia, Canada.

In Canada the traditional methods of harvesting are similar to Ireland, however, in recent years mechanical methods were introduced and this has resulted in some areas being overharvested. In 1985 a suction cutter was introduced and the percentage of material harvested by hand dropped to 20%. In recent years the proportion of hand harvested materials has gradually increased and now the majority is hand harvested. This was attained by government policy reducing the use of the mechanical cutter. The critical measure of sustainable use of the resource is the percentage of the landing which is made up of plants with holdfasts attached. Cutter rakes without sharp blades can have up to 15% of harvest with holdfasts attached and at very high exploitation rates (>80%). This increases up to 31% of the harvest removed by suction cutter. Nevertheless under normal operation less than 5% of the harvest by weight is made up of plants with holdfasts (Sharp, 1987).

Much of Nova Scotia was originally "Open ". Open areas require no harvesting plan nor do they have a restriction on the tonnage of *Ascophyllum* that can be harvested. Nowadays the resource is successfully managed (Sharp *et al*., 1994). The strategy adopted has been to allow companies exclusive rights to the resource. Granting of this licence is heavily dependent on the companies' ability to develop a harvesting plan and direct its harvesters (Sharp *et al*., 1994). Landings are reported at the level of sub-geographical areas. Overharvested areas have been closed to harvest to allow recovery. Harvesters submit daily logs of harvesting locations. The present *Ascophyllum* management plan for Nova Scotia is based on annual exploitation rate not exceeding 17% of the harvestable standing stock per sector.

Assuming that there is no fundamental difference between *Ascophyllum nodosum* in Nova Scotia and in Ireland the management would have to take into account the following factors; Monitoring of holdfast material cutting height and intensity, reliability of landing reports and environmental impact assessment (Sharp *et al*., 1994). A code of practice for harvesters should include: (a) cutting an area according to amount of standing stock available and (b) cutting at least an agreed level above the holdfast in order to ensure recovery of the resource.

LAMINARIA

A preliminary study of kelp resources is being undertaken at present and it is likely that these resources run into several million tonnes (Guiry, 1996). The most important kelp is *Laminaria hyperborea,* a subtidal species which exclusively colonises rocky substrata. Its upper limit is just below the level of low water at the lowest of tides and in clear water it can grow to a depth of 30 m or more. In Ireland, drift *L. hyperborea* is not cut but collected after spring storms on the coast (Guiry and Blunden, 1981). About 7000 tonnes of stipes and 2500 tonnes of blades (wet weights) (Guiry and Blunden, 1981) are gathered each year and exported to Scotland for alginate production. Collection is weather dependent and haphazard (Guiry, 1996). A preliminary desk study of the likely extent of the *Laminaria hyperborea* beds on the west coast has been carried out although further studies are needed to quantify precisely the actual resources and the effects of harvesting (Guiry, 1996).

Laminaria digitata is not harvested in commercial quantities but only small amounts are collected for food (Guiry and Blunden, 1981). This is despite the fact that *Laminaria digitata* is one of the largest seaweeds found along the European littoral, mature plants being 1-2 m in length (Guiry and Blunden, 1981) and it is probable that in excess of 100,000 harvestable tonnes is available in Ireland (Guiry, 1996). We currently have only a rough idea of the distribution of the resources of both species. Given the facts it would be surprising if this large, valuable resource remained untapped for much longer.

In Ireland at present, all *Laminaria* is collected by hand. However, this is likely to change in the near future. In France an apparatus known as the "scoubidou" is used extensively for harvesting of *Laminaria digitata* and this may be introduced for use here. It works by dropping a line down into the kelp and spinning it; the seaweed becomes caught. The line is brought up and the seaweed comes with it (Guiry, 1996). This method has the disadvantage of also removing the holdfasts. Harvesting kelp mechanically means that the whole plants are taken. Regeneration can be calculated from existing scientific data, but the difficulty is to apply data from the Isle of Man or Norway to Irish situations (Guiry, 1996).

In Canada, *Laminaria* is harvested by hand and by a method known as dragraking. This involves dragging a rake behind a tow boat. Residual biomass and plant density is a function of harvesting effort, gear selectivity and frond vulnerability. Vulnerability of a population is affected by bottom relief and frond size structure (Sharp and Pringle, 1990). Direct impact on the associated flora of kelp beds occur in their removal as a bycatch or by displacement of the substratum. The level of adverse impact on the substratum is low for dragraked harvests as the dragrake rarely touches the bottom (Sharp and Pringle, 1990).

Only drift *Laminaria digitata* and *Laminaria hyperborea* are collected, thus little or no management is required at present as this is natural wastage. However, should it be harvested mechanically on a commercial scale then the resource must be managed in a sustainable way. Prescriptions similar to those for *Ascophyllum nodosum* may or may not be appropriate. It was not found in the literature, to what degree the holdfasts and stumps versus new recruitment are responsible for the regeneration of the resource post-harvesting. Therefore harvesting impact and recovery studies must be carried out as they are critical for the sustainable development of the industry (Guiry, 1996).

RED ALGAE

Red algae of economic importance include *Palmaria palmata* or dulse and *Chondrus crispus*, *Mastocarpus stellatus* collectively known as carageen moss or Irish moss (Guiry, 1996). Our harvestable resources of carrageen moss are at most, about 300 tonnes and about 53 tonnes wet weight were gathered in Ireland in 1994 (Guiry, 1996). No survey of dulse has been carried out in this country, although casual observations indicate that it is commoner on the south and north coasts than it is on the east or west (Guiry, 1996). Dulse is a red alga that is eaten throughout the north Atlantic. About 9 dry tonnes are used in Ireland at present.

Chondrus crispus and *Mastocarpus stellatus* are bushy red seaweeds 6-15 cm in length which grow largely in the intertidal. They are gathered interchangeably and sold as Carrageen Moss or Irish moss. Collection is carried out from June to September on the coasts of Co. Clare, Co. Kerry, Co. Sligo and Co. Galway supplies more than 1000 tonnes of *Chondrus* wet weight per annum (Guiry and Blunden, 1981). They are collected by hand-picking and hand-raking. *Chondrus crispus* is found mainly in intertidal rockpools just above low water to just below high water. *Mastocarpus stellatus* is also found is rock pools but also is found in considerable quantities in the mid-intertidal of semi-exposed shores and is very rare in the subtidal.

Information relating to hand raking is limited. Pringle (1978) collected data from dragrakers and handrakers to observe aspects of the ecological impact of *Chondrus crispus* harvesting in eastern Canada. He generally noted the impact of handraking to be less ecologically damaging than for dragraking. For example, he found that the percentage of plants with holdfasts still attached in the harvest of handrakers was significantly less than those collected by dragrakes. He suggests that this could be due to the lightness and precision of the handrake. Hand-harvesters accurately direct the harvesting tool to the target species, thus ecological impact on associated species is minimal (Pringle and Matheieson, 1987). Furthermore fewer immature plants were seen in the harvest of handrakers than in that from any of the dragraking techniques. One reason for this is that holdfasts bear more immature than mature plants (Pringle, 1978). In contrast with the mechanised dragraker method, the state of maturation of the bed can only be judged by yield. In contrast, his results also suggest that handrakers harvest more immature plants which are independent of holdfasts. These plants could otherwise contribute to next years' biomass, however, this may be preventable by increasing the distance between the tines on the rakes (Pringle, 1978). Following a harvest impact study in Nova Scotia, a minimum legal tine spacing of 5 cm on handrakes was instituted (Pringle and Matheieson, 1987). In Nova Scotia's Northern gulf of Maine *Chondrus crispus* is harvested by hand using hand rakes. With this method holdfast removal, and thus retention of harvestable immature fronds was observed to be minimal (Pringle and Matheieson, 1987).

Most of the data found referring to the impact of harvesting dealt with solely mechanical methods of harvest. This is worth considering as it may be reality in Ireland soon. Pringle and Semple (1988) looked at harvesting of *Chondrus crispus* by dragrakes in the Gulf of St. Lawrence, Canada. The beds have been harvested here annually since the 1950's. However mean annual yield declined between 1972 and 1979 from the peak years 1966-71, through a standing crop decrease rather than harvest effort. Pringle and Semple (1988) noted that as the frond density and the biomass peaked in late June-July that increasing the yield would be as simple as moving the opening the date of the harvesting season from June 10th to July 1st. Furthermore they suggest that such intense harvesting over a period of generations could have made the present population develop characteristics to make them less susceptible to the rake. In Sharp and Pringle's 1990 review they looked at the ecological impact of marine plant harvesting in the north-west Atlantic. Although it appears that overall reproductive capacity of *Chondrus crispus* is reduced by intense dragraking, evidence to demonstrate recruitment overharvesting is lacking. There is little doubt there is a reduction of spore production, but how many spores are required to maintain desired yields? Indirect evidence suggests a shortage of spore production.

OTHER ALGAE

Two other brown algae of interest are *Alaria esculenta* and *Himanthalia elongata*. The growth and harvesting potential is being examined in *Himanthalia elongata* at present. *Alaria esculenta* is a large, kelp-like algae that grows on exposed shores. Irish plants grow to considerable sizes, up to 6 m in some areas and there is research into hybridising them with the Pacific species which grow up to 18 m in length. and cultivating this species (Guiry, 1996).

Seaweed aquaculture is in the very early stages. The only commercial seaweed aquaculture is in France where wakame (*Undaria pinnatifida*) is being grown for food and the European market is more than adequate(Guiry, 1996). In 1995, it was reported that the seaweed growing experiments were carried out at Carna Shellfish Research Laboratory. The main seaweed of interest is *Alaria esculenta* and work will soon start on *Gracilaria* which is the main seaweed used to feed abalones in Taiwan (Hensey, 1995). The mechanisms for cultivating this seaweed are well established (Kain-Jones, 1991).

The impacts resulting from algal culture would be expected to be minimal as they make their own food and excrete no waste products. The main concern would be if a non-native species was introduced this could have serious ecological and possibly genetic implications.

IMPACTS OF SEAWEED HARVESTING ON OTHER ORGANISMS.

Marine algae fulfil an important role for many marine organisms e.g. lobster, and as primary settlement substrates for some important shellfish species, as well as providing egg-laying sites for commercially valuable species (Caddy and Fischer, 1984). They also provide a substrate for spat to attach to, shelter from predation and act as food for grazing organisms such as sea urchins. Their removal is likely to negativly affect recruitment and survival in these species.

Ascophyllum has an associated assemblage of macroinvertebrates and is periodically utilised by fish and bird species (Sharp *et al*., 1994). Macrofauna most closely associated with the *Ascophyllum* would be reduced in proportion to the reduction in algal biomass. Harvesting it results in a bycatch of canopy invertebrates particularly the periwinkle *Littorina obusata.* However, according to Sharp *et al*. (1994) this represents only a removal in the annual harvest of 0.1% to 0.8% based on average canopy densities and by catch rates. Bodkin (1988) looked at the effects of kelp forest removal on associated fish assemblages in central California and found that the abundance of seven species of fish declined following kelp removal.

Although there is information available on organisms associated with algae (Baardseth, 1955; Pringle and Matheieson, 1987; Hickman, 1992), few experiments have looked at the effect of harvesting on these organisms. Studies have been carried out on the direct impact of harvesting *Chondrus crispus* on lobsters. Unexpectedly it was noted that average annual lobster landings in an intensively harvested area had increased by 86% over the previous ten year period; by contrast in a non-dragraked area landings only increased by 34% (Pringle and Matheieson, 1987). However this scenario is unlikely to be representative of organisms associated with seaweed beds.

SUMMARY

1. The following seaweeds are harvested in Ireland: *Ascophyllum nodosum* , *Laminaria digitata, Laminaria hyperborea, Chondrus crispus* and *Mastocarpus stellatus.*
2. All of the harvesting is carried out by hand. This is likely to change to mechanical harvesting in the near future and to have subsequent knock-on ecological impacts.
3. There is no management of the resource.

MANAGEMENT

The Irish seaweed industry is currently sustainable as nowhere near the full extent of our resources is being harvested. However, in order to maintain a sustainable industry we need to quantify the resource in each instance. This information is only available for *Ascophyllum* (Guiry, 1996). However, as well as knowing the standing stock of each of the seaweeds of interest we must also know what its annual production and its turnover time. Simple strategies can result in wise use of and sustainable management of the resource.

1. Non removal of holdfasts and cutting above a certain height on the plant when harvesting *Ascophyllum*, *Chondrus crispus* and *Mastocarpus stellatus.*

2. Harvest time and frequency are very important factors in the sustainable use of the resource.

Other organisms collected intertidally

SEA URCHINS

The value in size of the sea urchin fishery in Ireland has dropped overall from 94.6 tonnes in 1990 to 33.6 tonnes in 1994. This is indicative of a countrywide trend in sea urchin decline. The purple sea urchin (*Paracentrotus lividus*) has been collected on the Irish shores to satisfy mainly a French market demand. This has been well documented in Lough Hyne. Kitching and Tain (1982) counted 15,530 individuals in the south basin in 1979 whereas in 1995 a total of 15 individuals were recorded in the Lough by the same method (Maguire, 1995). This represents a decrease to 1.3% of the population from 1979 to 1995 (Maguire, 1995). However, like all natural populations the population of urchins in Lough Hyne is susceptible to natural fluctuations. The cause of decline may be attributed to a number of factors. The decline of the population due to overfishing around the north European and Irish coasts has been well documented (Maguire, 1995) and may also be due to some natural factor.

The loss of this key species will have serious ecological consequences for others. The predators of this species e.g. the starfish *Marthasterias glacialis* will eventually fall (although this is not the case in Lough Hyne at present). In addition the organisms that *Paracentrotus* prey on will increase in numbers. This is evidenced in Lough Hyne by the dense blanket weed present in the Lough in the summer of 1995. Maguire (1995) suggests that the cover would not have been as extensive if the numbers of urchins had been at their previous levels. This change will ultimately affect other species and may threaten the biodiversity of the Lough. Furthermore with the dramatic sea urchin decline, bathers now find this area attractive and so their increased use of the area is likely to result in further changes in the ecology.

MUSSELS

Mussels are collected around much of Ireland by hand and there has been nothing written on the impact of this activity. It would be expected that collection of mussels in the intertidal zone would compete with birds such as the oystercatcher. However the impact will depend on the scale of the operation. Hand pickers without commercial interests are likely to do minimal damage.

PERIWINKLES

There is a substantial trade in periwinkles with 3000 tonnes picked in 1995 according to the Department of the Marine. This appears uninvestigated, unmanaged and undoubtedly has an ecological impact on the species in the same food web.

Management of intertidal resources

Currently there is lack of management of Irelands intertidal resources. Generally no harvesting licences are needed and quantities removed are totally dependant on the operators. In order to manage these resources wisely we should:

- commission studies on the quantity and annual production of these resources.
- set quotas based on these findings.
- issue licences with a requirement to return the amount harvested.
- Investigate methods of harvesting which ensure a new standing crop in the least possible time.
- Produce a code of practice in consultation with the harvesters and other interested groups.

CONCLUSIONS

The aquaculture industry in Ireland has grown substantially over the past 10 years and is continuing to do so. What is needed is an overall plan for aquaculture so that our total coastal resources are managed in a sustainable manner. Ideally we want an industry which will create long term employment without compromising the future of the industry itself or the environment that it so closely relies on.

There are three key strategies to obtain sustainable aquaculture and intertidal harvesting these are:

1. An overall vision.
2. A comprehensive, efficient and effective licensing system for aquaculture and all other harvesting activities.
3. Consultation and cooperation at all levels with the harvesters and farmers, their representative groups.

1. An overall vision.

The best strategy for overcoming conflict between aquaculture and other water users is through clear environmental planning by the government at an early stage (Joyce, 1992). This plan should embody a vision of how the aquaculture industry will unfold in the coming years. Many of our best coastal sites are under threat and some have suffered long-term or permanent damage. We need a plan that will not only cater for continued expansion of aquacultural, and indeed intertidal harvesting activities, but wildlife conservation and other user groups.

2. The licensing system

Licensing is one of the key regulatory mechanisms. Unfortunately the licensing situation for aquaculture has had a troubled history. A series of out-dated and unworkable acts has left a significant proportion of the industry (notably the shellfish sector) unlicensed. The current situation has led to serious difficulties even knowing what activities are operational (Heffernan, 1995). Fortunately this has been recognised and steps are currently underway to revise the licencing procedure and legislation (BIM, pers. comm.). Ideally this new licencing system will incorporate a feedback system in order that the size and the locations of each enterprise is easily identifiable. A geographical information system would enable the position of each farm, their production capacity, disease history and any site information such as phytoplankton turnover rates or conservation status to be easily accessed.

3. Consultation

In order be successful in our endeavours to obtain a sustainable industry we must take a bottom up approach. We must educate and work with farmers and harvesters. Sustainable farming and harvesting will inevitably be more profitable (Soley *et al.*, 1992). Less food wasted equals more profit, less accumulated food on the sediment means less probability of outgassing of hydrogen sulphide, siting of farms away from seal haul outs or roosting sites for cormorants will result in fewer losses to predators. Harvesting sustainably will ensure that the crop is there to be harvested again in the shortest time period, thus ensuring maximum profitability. Farmers' attitudes are essential to a sustainable industry and every effort must be made to work with them in developing sustainable farming methods and informing them of the latest developments. Much consideration should be given to the subject of how to motivate farmers to farm in this way.

In order to adequately protect our environment from the negative aspects of mariculture we must consider how we define unacceptable change and how can we prevent it. In other words how much is too much? This limit must be decided *a priori* for water pollution, chemical dispersion, phytoplankton depletion, intertidal habitat occupation etc. To date, we have not even defined what we consider to be an unacceptable impact of a fish farm. This must be decided on. These are difficult criteria to define and they change from site to site. We must be conservative in our outlook as, for example, if bird numbers fall at a designated site (SPA) we are failing to meet our obligations under EU law. For some activities we know a great deal of the environmental impacts and for others the literature is filled with speculation and devoid of facts.

Where there is scientific doubt about the approval of a licence application we must err on the side on conservation particularly in our Special Protection Areas and Marine Nature Reserve. The onus must be on the farmer to show that the site is suitable for aquaculture as it is the farmer who will profit if the venture is successful. This principle is operating for finfish farms over 100 tons under the directive on environmental impact assessment. It is my belief that the principle should be extended to other mariculture activities. This stance is essential to maintain our wildlife at current numbers and to protect the international reputation of our aquaculture industry. It is in the interest of the aquaculture industry to be, and to be seen to be, operating in harmony with the environment. An internationally recognised environmental quality standard is necessary in order to give credit and greater marketability to those aquaculture operations which operate in a sustainable manner.

For a country that has so much invested in aquaculture, too little is invested in research. Research should be supporting the industry in finding cost-effective methods of operating with minimum environmental impact. At present Ireland has a booming economy and many of our resources are being realised. However, there is a limit to growth. The aquaculture industry must find its limit. Because the industry is so new this limit is undefined at present. Perhaps it will be defined in terms of percentage of intertidal zone occupied by shellfish culture or millions of gallons of water or hectares of seabed per fish farming operation. Research will help us to find these answers.

Recommendations

Bord Iascaigh Mhara have made the first steps towards building a sustainable industry in Ireland by appointing an environmental officer. In addition, they have surveyed the industry and collated codes of practise based on industry experience to date (BIM, 1996; 1997). As would be expected the main trust of these codes of practise are quality and profitability. I would recommend that these be used as a starting point, in conjunction with the findings of this report, to produce environment oreintated codes of practice for use by the industry.

A further recommendation is that funding be made available to educate the farmers about the impacts of fish farming along with the legal implications of the habitats and birds directive. Education should be aimed at empowering them to find their own practical and environmentally aware solutions to problems they may encounter at their sites. Above all the farmers themselves must be motivated, be it for profit or otherwise, to farm in a sustainable manner.

It is envisaged that management agreements will have to be reached with these farmers as part of the design and implementation of plans for nature conservation that is required under the habitats and birds directives. Therefore it is recommended that good relations are developed at all levels from the boardroom to the the mudflats. A spirit of cooperation should be fostered between the farmers and local wildlife rangers and between executives of the NPWS, BIM and DOM. Ultimately these relationships will be critical to the successful implementation of any management plans and to the development of this industry in harmony with nature.

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APPENDIX 1.

In order to find the relevant papers for this literature review the following species strategies (Table 1) were combined with each topic strategy (Table 2). (* signifies no defined continuation of this word - allows single and plural to be included).

Table 1; Species strategy

SPECIES		STRATEGIES
Finfish		
Salmon	*Salmo salar*	SALMO* AND SALAR
Trout	*Oncorhynchus mykiss*	(SALMO* OR ONCO OR ONCHO) AND MYKISS
Turbot	*Scophthalmus maximus*	SCOPHTHALMUS AND MAXIMUS
Halibut	*Hippoglossus hippoglossus*	HIPPOGLOSSUS
Shellfish		
Oysters	*Ostrea edulis*	OSTREA AND EDULIS
	Crassostrea gigas	CRASSO* AND GIGAS
Clams	*Ruditapes philippinarum*	(TAPES OR RUDITAPES) AND (PHILIPINAR* OR PHILLIPINAR*)
Mussels	*Mytilus edulis*	(MYTILUS OR MYTILLUS) AND EDULIS
Scallops	*Pecten maximus*	PECTEN AND MAXIMUS
Abalone	*Haliotis tuberculata*	HALIOTIS AND TUBERCULATA
	Haliotis discus hannai	HALIOTIS AND DISCUS
Sea Urchins	*Paracentrotus lividus*	PARACENTRO* AND LIVIDUS
	Psammechinus miliaris	PSAMMECHINUS AND MILIARIS
Periwinkles	*Littorina littorea*	LITTORINA AND LITTOREA
Crustacea		
Lobsters	*Homarus gammarus*	HOMARUS AND GAMMARUS
Algae	*Alaria esculenta*	ALARIA AND ESCULENTA
	Ulva spp.	ULVA
	Gracilaria spp.	GRACILAR*
Intertidal Harvesting		
Lugworms	*Arenicola marina*	ARENICOLA AND MARINA
Ragworms	*Nereis virens*	NEREIS AND VIRENS
Seaweed	*Laminaria*	LAMINAR*
	Ascophyllum	ASCOPHYLL*
	Palmaria palmata	PALMARIA AND PALMATA
Cockles	*Cerastoderma edule*	CERASTO*
Sea Urchins	*Paracentrotus lividus*	PARACENTRO* AND LIVIDUS
	Psammechinus miliaris	PSAMMECHINUS AND MILIARIS
Periwinkles	*Littorina littorea*	LITTORINA AND LITTOREA

Mussels	*Mytilus edulis*	(MYTILUS OR MYTILLUS) AND EDULIS

Table 2. Topic Strategy

Environment	ENVIRO*
Ecological	ECOL*
Impacts/Effects	IMPACT* OR EFFECT* OR INTERACTION
Intensive culture	INTENSIVE CULTURE
Extensive culture	EXTENSIVE CULTURE
Water quality	WATER QUALITY
Pollution/Toxicity	POLLUT* OR TOXIC* OR EFFLUENT*
Waste	WASTE*
Feed/food	FEED* OR FOOD
Oxygen	OXYGEN*
Nutrient	NUTRIENT*
Nitrogen/Ammonia	NITR* OR AMMONIA
Phosphorus	PHOSPH*
Sulphate	SULPH*
Suspended matter	SUSPEND* OR TURBIDIT*
Organic matter	ORGANIC
Sediments	SEDIMENT*
Chemicals	CHEM*
Additives	ADDITIVE*
Antifoulants	ANTIFOULANT*
Pesticides	PESTICID*
Antibiotics /Antibacterial	ANTIB*
Biota/Benthos	BENTH*
Introduced species/Exotics	INTROD* OR EXOTIC*
Escapees	ESCAPE*
Disease	DISEASE*
Predator/predators	PREDAT*
Birds	BIRD*
Seals	SEAL*
Wildlife	WILDLIFE
Management	MANAG*
Dredging	DREDG*
Hydrology	HYDROL*

Intertidally harvested

Environment	ENVIRO*
Ecological	ECOL*
Habitat	HABITAT*
Intertidal	INTERTID*
Impacts/Effects	IMPACT* OR EFFECT* OR INTERACTION
Bait	BAIT

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