CASE STUDY 6.4

RISK ASSESSMENT OF THE DECLINE OF MACROPHYTES DUE TO MARINE FISH FARMING WASTE MATTER

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6.4.1 Introduction

In this case study, we assess whether hazards released from fish farms could adversely affect adjacent seaweed beds, in particular, kelp beds. This exercise was carried out to identify potential hazards and assess risks. This assessment can be used as an example for carrying out assessments on other forms of macrophytes and marine plants. The analysis starts with the identification of the issue(s) of concern.

6.4.1.1 Introduction and Background Material

Kelps are the largest macroalgae and are members of the order laminariales. They are predominately benthic macroalgae of cold water, and occur throughout the salmon farming areas of Europe and North America where they are valued ecologically and economically in some regions. In Norway, which is one of the largest salmon farming producing countries, wild *Laminaria hyperborea* is harvested and used for the extraction of alginate. Harvesting is restricted to monotypic *L. hyperborea* beds that are only found in completely exposed sites. Currently such sites are unsuitable for salmon farms that must be sited, for structural reasons, in less exposed areas (for example, within a fjord instead of along a completely exposed coastline). Ecologically, a large natural kelp bed protects against coastal erosion (Madsen *et al.* 2001), many animal and algal species are associated with the beds, some fish and bird species feed directly on them, and some fish spawn directly on the fronds. In one study, for example, 387 species were found on its holdfasts (Moore 1971). In a more recent study, 238 species of mobile macrofauna with an average density of almost 8,000 individuals per kelp were found on the *L. hyperborea* sampled along the Norwegian coastline (Christie *et al.* 2003). The largest numbers were found on the largest specimens. Kelps have declined around the world due largely to over fishing of the predators of their major grazers, and excessive sedimentation and nutrients supposedly promoting shading by microalgae (Steneck *et al.* 2002). The areas where kelp beds have completely disappeared due to sea urchins are called urchin barrens (Sivertsen 1997). Over fishing the sea urchin predator has been the suggested cause of the large number of these barren sites, and is thought to be the greatest manageable threat to kelp forest ecosystems over the 2025 time horizon (Steneck *et al.* 2002).

Marine fish farm waste effluent consists of particulate matter as well as dissolved nutrients (Figure 6.4.1). Each fraction has the potential to affect kelp. For example, over accumulation of particulate matter from the fish farms in the benthos leads to an increase in the anoxic zone and over production of sulphide, which is toxic at low concentration to macrophytes and marine plants. As well, young kelp can be buried. Around the world coastal anthropogenic dissolved nutrients input especially nitrate inputs have simulated or over simulated algal growth, especially in areas where background levels limit growth.

The ecological importance and the decline of kelp beds necessitate their protection. From the above information, it is clear that waste matter from coastal marine fish farms is a hazardous substance to kelp (for example, the effluents pose a threat to the health of kelp). But, what is the effect on a local population? To answer this question, we developed a risk assessment model, and tested it on two hypothetical farming sites. The logic model that was developed is conceptualised in summary form in Figure 6.4.2. The model consists of release, exposure and consequence assessments, and risk estimation. The process starts with the question ‘is there kelp habitat in the area of interest’. The endpoint to the overall assessment is an estimation of the net change in kelp population. If the answer to the first question is ‘yes’, the next question to be answered is ‘is there a fish farm nearby’. If the answer to this question is ‘yes’, then we question whether kelp is growing nearby. If the answer to that question is ‘no’, then the area is possibly barren or once kelp grew there. Note, at this stage of a scoping exercise, it is assumed the hazard did not cause the barren. To be complete, the assessment would, however, have to address the possibility that fish farming effluents are a hazard to kelp recovery as well as to an existing kelp bed. The possibility of a barren site is flagged so that recovery can be considered in risk mitigation. The next stage is a release assessment, followed by an exposure assessment, and culminates in a consequence assessment. In the consequence assessment the severity, intensity and duration of the hazard, exposure and exposure consequence are estimated for each hazard using a logic model. The logic model is developed via analysis of available literature, measurements and/or modelling exercises. The consequence assessment is based on a summary of information derived from experience and as such it describes the likely outcome under ‘average’ technological and environmental conditions. Each new site may have special regulator restrictions or apply new technologies which may modify the anticipated affect.

Figure 6.4.1 : Conceptual diagram of the pathways of effects of fish farm waste effluents.

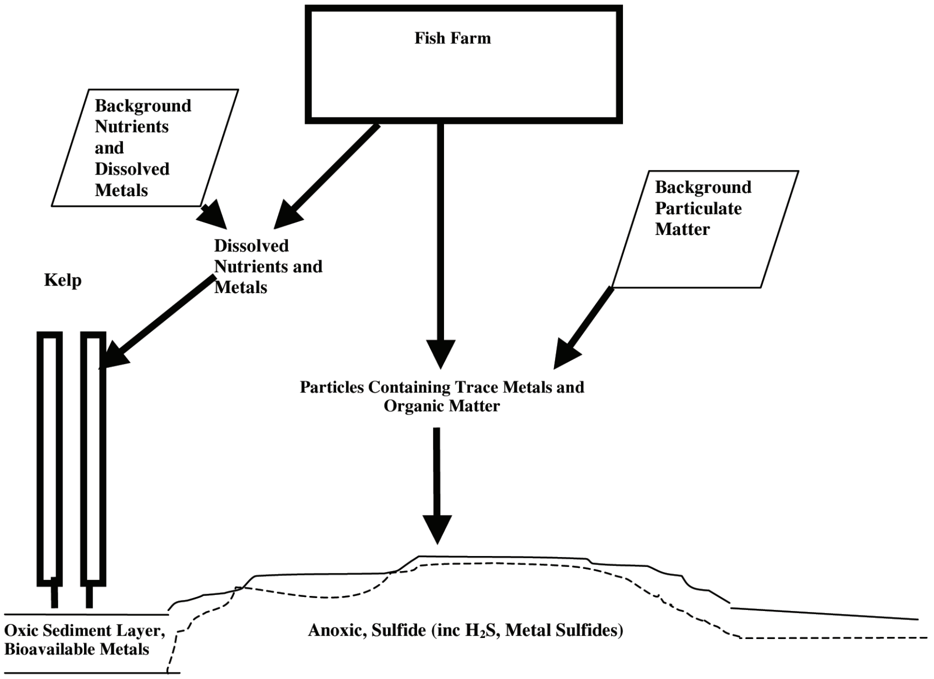
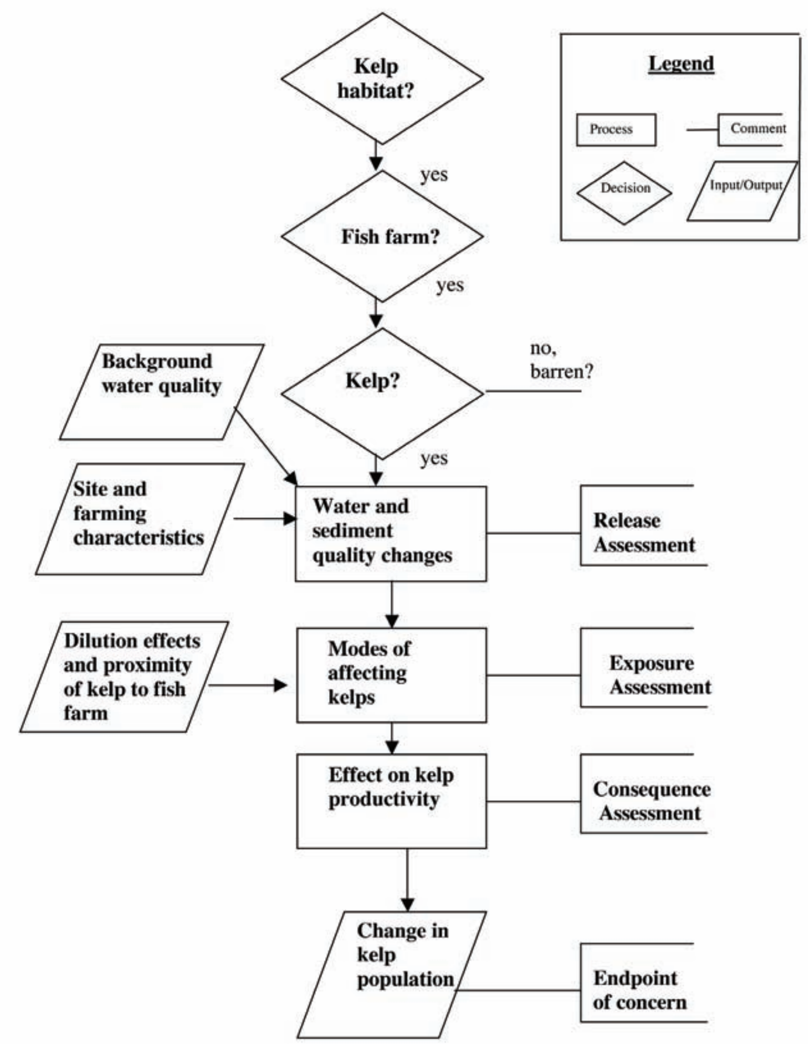


Figure 6.4.2 : Conceptualised logic model used to determine the risk of kelp decline due to fish farming waste.



The risk estimation step in the assessment allows the assessment to modify the nominal outcome accordingly. Risk mitigation, risk communication and management, and how to apply the assessment to other macrophytes are discussed near the end of the document.

Risk management has been defined as a set of activities which if carried out lessens the impact of hazardous substances. We used the term to describe only those activities that follow the risk assessment. The analysis begins with a description of the two fish farm sites, and the kelp life history and basic biology. The latter is used to determine exposure pathways, and describe and quantify the relevant conditions of kelp to risk agents.

6.4.1.2 Site description

For this exercise, we created the two hypothetical fish farming sites in the Lofoten to North Cape region in Norway. This area was chosen because key information such as kelp population, water quality, flushing conditions and salmon farming are available, and back ground water quality is good. Farming sites in Southern Norway could have been examined, but as water quality there is affected by municipal waste, agriculture and industry, it would have been more difficult to separate the effect of just salmon farming waste on kelp. The two fish farms reside at approximately 69 0N and 15 W (Sites A and B, Figure 6.4.3) and within well studied Laminaria resource areas (Sørum *et al.* 1990; Sivertsen and Hopkins 1995; Sivertsen 1997; Rinde and Sjøtum 2005). They are 5 km apart with connecting water currents within a fjord. Table 1 contains a description of the farming sites as taken from Carroll *et al.* (2003) including the production level described using annual fish feed amounts, condition of the substratum, and the current and depth under the cages. Site A is located in a sheltered site, while Site B is a more exposed and deeper site located nearer the fjord inlet. At the time of the study by Carroll *et al.* (2003), a Norwegian aquaculture permit allowed a higher fish production level at less environmentally sensitive sites which were deemed to have higher current speeds, so the fish production at Site A is less than at Site B. Both farms are located approximately 60 m from shore, and orientated parallel to it and the prevailing current. Kelp beds are present near both sites.

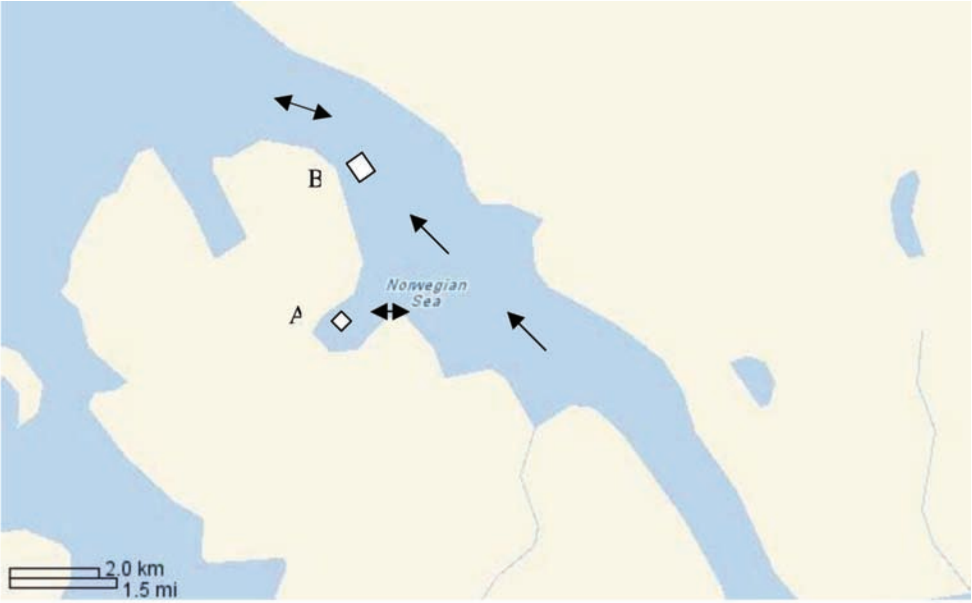
Within the Norwegian coastal region between Lofoten to North Cape, kelp around typical salmon farming sites (for example, sheltered and semi-exposed sites) are highly preyed upon by sea urchins. *L. hyperborea* have optimal growth rates due to favourable temperatures during the growth period and long summer days during the time of carbon production and storage (see section 6.4.1.2 Kelp life history and biology). Rapid growth tends to make them the largest in Norway (Rinde and Sjøtun 2005). The age of maximum growth is 4 years, and the average length is 1.5 to 1.61 m at a water depth of 5 to 8 m.

Coastal currents outside of the fjords in this area where the salmon farms reside move strongly up the coast, and in and out of the fjords (Figure 6.4.4). Within the fjord, surface water of low salinity arising from a river moves outward towards the open sea, while denser Atlantic water moves in below it (Aure and Skjoldal 2004). This is called estuarine circulation. In addition, wind-driven up and down welling just outside of the fjord causes considerable water exchange in the intermediate water layers. Usually basin renewal is yearly in the winter or spring when the density of the water outside of the fjord is at its seasonal maximum. Residence time for surface water and intermediate waters is on the order of days. The water quality in this area has been deemed excellent due to high flushing rates, the lack of large cities and small-scale industries with limited pollution potential (Aure and Skjoldal 2004). There, naturally occurring dissolved nitrogen in the form of nitrate is most available near the upwelling areas at the fjord inlets. Nitrate concentration increases with increasing salinity, and peaks within the Norwegian coastal current in that region in April at 8-14 M, while extreme lows occur in summer during times of high freshwater inputs (Aure and Skjoldal 2004). Salmon farming within the region contributes approximately 50 and 70% of the regional estimated total yearly N (approx. 10,000 tons) and P (1000 tons) inputs, respectively (Aure and Skjoldal 2004), and there are no major point sources of either nutrient. Due to the background nitrate cycle, absence of significant N point sources, and constant supply of N from fish farms, coastal marine fish farms within the area during the late spring and summer potentially provide macrophytes a source of nutrients normally not available to them.

6.4.1.3 Kelp life history and biology

In this case study, both *L. hyperborea* and *saccharina* are considered because they naturally grow together, and are dominant species in the area of interest. Much of their basic biology with the exception of more recent records pertaining to geographical distribution, predation, nutrient uptake and pollution effects is reviewed in Kain (1971, 1979) and Dawson (1966). These three older references are used within this document without further citation. Their life history and basic morphology are depicted in Figure 6.4.5. At maturity (two years for *L. hyperborea* and first summer for *L. saccharina*), the sporophyte (the large plant easily visible to the naked eye), consists of a blade or lamina and a simple cylindrical stipe attached to substrate by a hapteroid holdfast. This plant produces superficial sporangia on sporphylls or sori on its blades. Sori are produced in winter for *L. hyperborea*, and in summer for *L. saccharina* after the blades or laminae have fully developed. These mature in succession and fall out of the blades as irregular plates to end up in the benthos. Meiosis occurs in the developing unilocular (single cell) sporangia, which usually release 16-64 zoospores which attach after a brief period of motility onto solid rock, stones (> 25 cm), loose substrata such as gravel, dead shells, corallines and in the case of *L. saccharina* soft bottom as well (Schaffelke *et al.* 1996). Some species of seaweeds including kelp species are capable of attaching to nets, ropes and buoys on floating fish farms, and grow so densely as to foul them. After attachment, they germinate within a few hours to weeks to grow into tiny haploid gametophytes (Sivertsen and Hopkins 1995; Schaffelke 1996). The spore dispersal range for *L. hyperborea* is at least 200 m from a kelp bed (Sjøtun *et al.* 1995). The fertilized egg secretes a wall and soon undergoes division to begin the development of the sporophyte.

Figure 6.4.3 : Physical location of hypothetical salmon farming sites A and B within a Norwegian fjord.



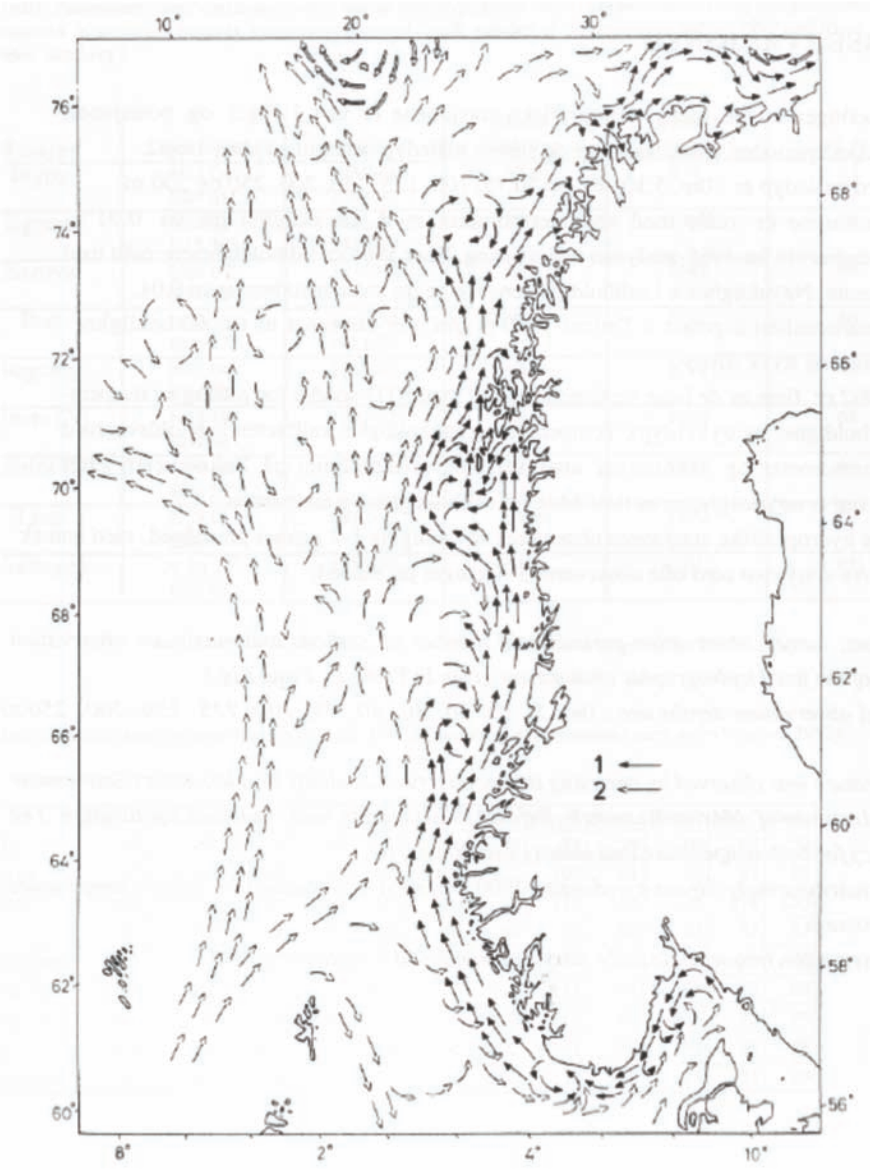
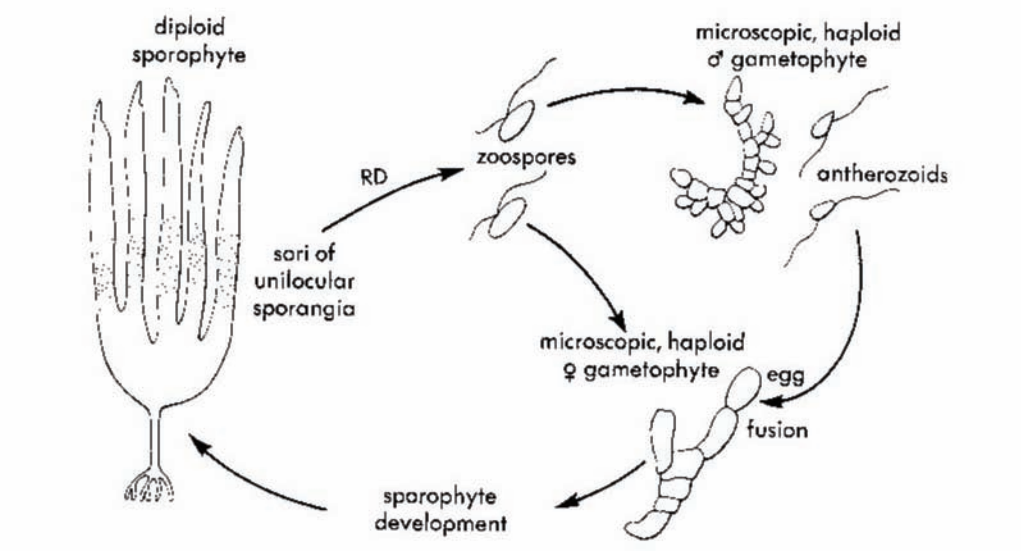


Figure 6.4.4 : Direction of current along Norwegian coastline and near fjord inlets (Aure and Skjoldal 2004, reprinted with permission). It is important to note how the current enters the fjords near our study area. The dark arrows indicated as number 1 in the diagram represents the Norwegian Coastal Current and the lighter arrows indicated as number 2 represents the Atlantic current. The degrees longitude and latitude are indicated on the border of the diagram.

Figure 6.4.5 : Diagrammatic life history of Laminaria *sp. (depicted here is* L. hyperborea*). From Dawson 1966. The sporophyte consists of lamina or blade connected to a holdfast by a stipe.*



Both species can grow at depths between low tide levels to 30 m at wave-exposed to sheltered sites, but the growing depth is actually limited by grazing, irradiance or substratum. Between species, growth patterns are distinct. For *L. hyperborea*, its holdfast and stipe can live 8-20 years, while the blades can break off in severe winter storms. From the stipe just below the old blade, a new blade emerges yearly. If the stipe dies, so does the kelp. The dark period between November and March is its major growth season when it uses C stores obtained during the summer well lit period (Lüning 1971). Sporophyte lengths is 1 to 2 m. Growth rate and mean standing density in shallow winter are approximately 0.94 cm/day and 11-10 kg/m^3, respectively. Average densities of *L. hyperborea* not impacted by harvesting or predation are 20.7 individual/m^2 and 23.9 individuals /m^2 for adult and juvenile specimens, respectively (Sivertsen 1997). The largest kelp originate from new blades that were able to use the nutrients in the blades from the previous year's growth.

In contrast, the holdfast of *L. saccharina* usually lives 3-4 years, while the blades and stipe tend to break off during winter storms. From the holdfast, new blades emerge yearly. If the holdfast dies, so does the kelp. The maximal growth period is between January and June instead of during the winter as for *L. hyperborea*. Ninety-three percent of the linear growth occurs in the proximal 25 mm of lamina during the first four months of the rapid growth period. Lamina length varies from 1 to 2 m. Growth and shallow-water biomasses are 1.7- 4.87 cm/day and 8-20 kg/m^3, respectively. The exception to this growth pattern is found in the high Arctic (74^o 18'N; 20^o 14' W) where blades live 2-3 years, plants longer than 4 years, and blades, as blades of *L. hyperborea* do, elongate during the dark period (Borum *et al.* 2002).

Background nutrient levels are important. In *Laminaria*, ammonia uptake saturation occurs at 10 µM while nitrate uptake has been shown to increase with background levels up to 60 M (Ahn *et al.* 1998; Chapman *et al.* 1978; Harrison *et al.* 1986). Both nitrate and ammonia are taken up simultaneously at nearly equal rates, if available. Chapman *et al.* (1978) observed a linear relationship between the growth of *L. saccharina* and NO-$_3$ concentration up to 10 µM and a luxury consumption of NO-$_3$ above 10 µM. Laminariales tend to store nutrients during periods of high nutrient levels to use for growth during periods of low availability (for example, Chapman and Craigie 1977).

The main competitor of *L. hyperborea* except near low water is *L. saccharina*. The outcome of this competition is often determined by the substratum. *L. hyperborea* prefers solid rock, and *L. saccharina*, due to its more flexible stipe, can out compete it if attached to substratum that can move in severe weather. In sheltered areas *L. saccharina* out competes *L. hyperborea* (although it can grow there), while generally *L. hyperborea* out competes *L. saccharina* in completely exposed areas. With decreasing wave exposure and current, *L. hyperborea* weakens. In shelter sites, the holdfast is poorly developed with few branched haptera on its holdfast and, as a result, it poorly attaches to rock. In addition, its stipe is short in sheltered sites, and the blade or frond is thin, brittle and easily torn. The morphology of *L. saccharina* also changes with wave and current conditions. Stressors typical of exposed environments (for example, breakage, dislodgement) are viewed as the cause of the differentiating morphological characters (Fowler-Walker *et al.* 2006).

A kelp population has the capacity to recover quickly from a major decline. This has been documented after they have been harvested. In Norway, for example, resource areas for kelp harvesting are large, as exemplified by the harvester size and speed which is 30 tons and 2 tons per minute, respectively (McHugh 2003). Harvesting areas are divided into 5 sections approximately 1.6 km long, and each section is harvested every fifth year after which new recruits would be 1-2 m high (Kitching 1941). The key to rapid recovery is harvesting only older individuals so that the younger individuals living in the under story can grow quickly in the light exposed by the harvest (Christie *et al.* 1998). Unfortunately five years is only a sufficient period for regeneration of the kelp canopy. It does not permit sufficient time for full regeneration of the organisms that once lived upon it (Christie *et al.* 1998).

Abundance of these two species is affected by over grazing by sea urchin (for example, Hjorleifsson *et al.* 1995; Sivertsen 1997); harvesting, sedimentation over young (Devinny and Voise 1978; Walker and Richardson, 1955); increasing water depth and/or light limitation (for example, Kain 1979; Gerard 1988), latitude (Rinde and Sjøtum 2005), current/wave action (Madsen *et al.* 2001; Sjøtun *et al.* 1993; Sjøtun *et al.* 1998); canopy biomass (Subandar *et al.* 1993; Sjøtun *et al.* 1998), nutrients (for example, Chapman and Craigie 1977), and pollution as well as other anthropogenic stresses (Coelho *et al.* 2000; Bellamy 1968; Bellamy *et al.* 1970; Nakahara 1973; Burrows 1971; Burrows and Pybus 1971; Hopkin and Kain, 1978). This information was used to help identify the risks associated with salmon farming waste on *Laminaria*.

6.4.2 Hazard Identification

A hazard produces risk only if an exposure pathway exists and the exposures create the possibility of adverse consequences. This section is used to identify possible adverse consequences of kelp exposed to fish farming effluents.

6.4.2.1 Possible effects of salmon farming effluents on seaweeds

Little research has been carried out on quantifying the effects of aquacultural effluent on wild plants and macroalgae. On the other hand, cultivated seaweed receiving aquaculture effluent has been the subject of many research investigations around the world. In one, Petrell and Alie (1996) showed that kelp cultivated on ropes adjacent to fish farms can act as biofilters of the dissolved nutrients in salmon waste matter. In one study focused on wild individuals growing within a lagoon system, macrophytes and macroalgae species composition changed with distance from the farming operation (De Casabianca *et al.* 1997; De Casabianca *et al.* 2003), and the overall species richness was low (Bachelet *et al.* 2000). These trends are consistent with changes in species composition and richness within a polluted kelp bed (Jones 1971; Bellamy 1968). In one polluted kelp bed, 92% of the kelp biomass disappeared along with a corresponding loss of associated fauna. In a closed lagoon containing shellfish farming, the macroalgae and macrophytes receiving additional nutrients in the effluents were viewed as vehicles for additional sources of dissolved nutrients once they had died and decayed (De Casabianca *et al.* 1997). The growth and N and P nutrient content of *Fucus vesiculosus L.* and associated biomass of epiphytes species were higher close to a fish farm than in a reference site in the archipelago of Aaland, southwest Finland (Roennberg *et al.* 1992). As well, towards the farm, the species composition of algal epiphytes shifted in dominance from brown and red to green algae. These examples provide a reason to examine the risks that nutrients and waste matter from salmon farming might engender for local stocks of *Laminaria*.

We determined after analyzing biological characteristics and other data that *Laminaria*'s growth pattern and reproductive cycle makes it susceptible to the build up of waste matter and dissolved nutrients. Specifically, a *Laminaria* population (numbers, age class composition, growth) is at risk if the waste matter from a fish farm contains high levels of dissolved nutrients that: a) promotes growth of mainly older *Laminaria* individuals, b) promotes growth of algal epiphytes that subsequently negatively affect kelp laminae or its function, and c) modifies the way nutrients are stored for a future generation. A *Laminaria* population is also at risk if the solid waste matter contains particulate matter to levels that cover substratum, gametophytes, holdfasts or blades, or cause a reduction in photosynthesis due to high turbidity. In addition, the population is at risk if the waste matter contained high levels of toxic material. In our analysis, we carefully considered early life stages as a recent review of the anthropogenic effects on the early development stages of seaweeds stresses the importance of the early life stage with regards to the viability of a seaweed bed (Coelho *et al.* 2000). The possible affects of increased nutrients and solid waste matter on sporophyte growth, sori production and regeneration in the benthos that we found are described within this section. The methodology to produce a quantitative measure of the risks is discussed under risk estimation.

6.4.2.2 Possible adverse effects of dissolved nutrients on kelp growth and population

Laminaria growth is highest under low-medium kelp density, adequate nutrients, and mid velocities or level wave action, as under these conditions competition for light and nutrients is low, and nutrient exchange with the laminae is enhanced (Subandar *et al.* 1993; Sjøtun *et al.* 1998; Madsen *et al.* 2001; Hurd *et al.* 1996). An older established bed retards current, restricts light (especially to lower canopy seaweeds) and is more effective due to its larger biomass at removing nutrients to low levels. A high biomass level restricts growth in *L. saccharina* during its rapid growth period and/or the period when nutrients are available for growth and storage. Similarly, high biomass levels during the well lit period could also reduce growth in *L. hyperborea* during its high growth period or winter when it depends on stored carbon from the well lit period for energy.

In general, kelp biomass increases with an increase in nutrient availability to a level where self-shading of the upper story population occurs, shading of younger and smaller individuals in the low stories occurs and/or restricting of flow and nutrient exchange throughout the bed occurs. Light limitation and current reduction have been suggested as the reasons why growth in kelp beds consisting of younger individuals without an upper story of older individuals is higher than the growth in mixed aged beds (Sjøtun *et al.* 1998). In that study, growth of 2-

3 year old individuals declined by half as lamina biomass doubled (from approximately 300 to 600 kg/m^2).

Epiphytes normally grow on the stipe, holdfast and older blade tissue of laminariales. Epiphytic algae type and abundance change due to fish farming (Roennberg *et al.* 1992). Williams and Ruckelshaus (1993) observed that the growth of eelgrass (*Zostera marina*) decreased as epiphytic algae increased. At greater than 80% ambient nutrient levels, epiphytic algae on kelp and eel grass increased relative to background levels (Russell *et al.* 2005). A five hour 200 µmol l^{-1} pulse of nitrate caused an increase in epiphytic algae and a corresponding decrease in growth in the host, while a series of shorter-duration pulses produced no differences (Worm and Sommer 2000). These researchers suggest that the duration of the nutrient pulse and nutrient prehistory on epiphytic algae growth appears to be more important than the nutrient concentration. In summary, the level and duration of nutrient pulses can affect lamina size and function due to an increase in epiphytic algae and salmon farming effluents have been shown to change the type and abundance of epiphytic algae, but research is lacking to show that such changes cause a decline in a local kelp population.

Holdfasts and laminae have been shown to store nutrients for future growth and a new generation, and typically these nutrients are obtained in the early spring in temperate coastal areas when light levels are low and nutrients levels are high. Nutrients stored in the holdfast are obtained from covering soft sediment and passed up to the laminae when needed (Williams 1984; Raven 1981). For nitrate uptake during the spring upwelling period, stored carbon is often used for energy (Raven 1981; Lüning 1971). Typical nutrient uptake and storage characteristics of kelps may, however, be altered in seasonally unusually high nutrient loading conditions. First year *L. saccharina* grown in tanks receiving fish farming effluent under high irradiance did not store N in the blades (Subandar *et al.* 1993). Ahn *et al.* (1998) found that the nutrient uptake kinetic parameters of first year kelp that had originated from a salmon farming net cages were within expected ranges. Like in the previously mentioned study nitrogen did not store in the lamina as a result of the fertilization. In another study, nutrient depleted Laminaria was fertilized and growth was enhanced, but C reserves did not build up in the usual matter (Chapman and Craigie 1977). The possibility exists that the lack of nutrient/carbon reserves in kelp grown in these ways weakens the next individual arising from the remaining tissue from the parent seaweed (Chapman and Craigie 1977). In summary, evidence suggests nutrient fertilization changes the nutrient/energy storage pattern of kelp relative to unfertilized kelp, but as yet, research has not been conducted to determine if such changes could cause a decline in a local kelp population.

The effect of reduced biomass production of sporophytes, whether due to epiphytes, nutrient storage conditions, light limitation, etc., could affect future generations of *L. hyperborea*, as strong new growth on this perennial depends largely on the condition (of energy and/or nutrient stores) and size of the holdfast, stipe or blades (depending on the species) generated from the previous year growth. In addition, adequate growth is important for adequate sori production. Gametophyte and young sporophytes viabilities are considered generally poor so ample sori production is important for Laminariales as a way to ensure future generations. As was previously mentioned, sori production occurs after blades have elongated near the end of the growth season. Thus factors that were mentioned within this section that affect lamina growth and condition would also affect sori production.

6.4.2.3 Possible adverse effects of particulate matter on kelp

Salmon farms using cage technologies produce particulate matter consisting of carbon, nitrogen, phosphorus, sulphur, and trace metals. The particle size ranges from 0.075 to 25 mm for salmon ranging in size from 1 to 5 kg (Buryniuk *et al.* 2006). This matter, along with uneaten fish feed degrade into smaller particles due to current forces and bacterial action. According to basic sedimentation theory and assuming equal particle density for all particle sizes, the largest particles settle out under or close to a fish farm, while the smallest particles settle further away or even remain suspended. The benthic community changes when the degradation rate in the benthos proceeds at a lower rate than the input rate. Through the process of decomposition, oxygen in and above the sediments can become depleted, and under anoxic conditions gases such as nitrogen, carbon dioxide, methane, and hydrogen sulphide can be generated. Direct effects of fish farming particulate matter on natural kelp beds have not been studied. Research has, however, been carried out on other forms of debris and under different cultivation conditions:

Sedimentation over kelp beds has been suggested as the cause for a decrease in the maximum depth of kelp in southwest Ireland, although the causal mechanisms were not known at the time of the study (Edwards 1980). Sedimentation has been shown to affect the development of gametophytes. A small layer of sediment or the amount to just cover the surface of a culture plate reduced the survival of gametophytes of the giant kelp, *Macrocystis pyrifera*, and 108 mg/cm^2 was sufficient to kill them. Water motion further decreased viability (Devinny and Voise 1978). Fletcher (2002) found that fine sediments just 3 mm thick, significantly reduced survivorship of *Fucus* embryos by restricting diffusion of metabolic waste products.

Sedimentation can harm *L. saccharina* if holdfasts are buried to the extent that new stipes and laminae can not emerge. Prior to burial to this level, the kelp would have, however, have already encountered the toxic effects of sulphide (*see* next section). Sediment depositing on top of kelp blades to the level that restricts photosynthetic activity is another avenue of potential harm. Some evidence supports this possibility; tanks containing kelp and receiving fish tank effluents had to be cleaned every two days to remove the debris on walls and laminae (Subandar *et al.* 1993). Sedimentation to this degree (in still water) is, however, unlikely in wave swept environments typical of coastal waters and salmon farms.

Sedimentation in some polluted areas has been sufficiently severe to cause turbidity, light limitation and decline in kelp beds (Bellamy *et al.* 1970; Edwards 1980). Under low to mid current velocities, high kelp biomass can potentially aid in increasing sedimentation rate and reducing turbidity (Madsen *et al.* 2001). Salmon farms are not associated with producing overly turbid water, although, they can be sited within waters that are seasonally quite turbid with Secchi disc reading of approximately 1 m (Ang and Petrell 1998).

In summary, waste particles have the potential to affect seaweeds especially if the particles settle over preferred substratum containing gametophytes.

6.4.2.4 Potentially toxic substances associated with salmon waste

Brown macroalgae are widely noted for their ability to accumulate dissolved potentially toxic heavy metals such as Cu and Zn commonly found in salmon farming effluents, other waste streams and naturally in the marine environment by 10 to over 100 times background levels without known biological effects in adult specimens. The amount accumulated tends to be directly related to growth rate and the concentration in the water (Lobban and Harrison 1994). In a one year old brown alga, *Ascophyllum*, collected on rocks near salmon farms with and without nets, treated with copper as a biofoulant, copper levels varied between 3.4 to 8.7 mg Cu/ kg, and the values for the one non-copper treated site fell to the low range (Solberg *et al.* 2002). The higher values are typical of *Fucus* sp. and *Chondrus crispus* collected near a steel plant wharf (Sharp *et al.* 1988).

Different dissolved heavy metal solutions are more toxic than others (*see* Lobban and Harrison (1994) for information on the factors affecting and mechanisms of metal toxicity). For example, Cu is more toxic to *L. hyperborea* than Zn (Hopkin and Kain 1978). Typical oceanic Cu concentrations need to increase by a factor of three to be toxic to *L. hyperborea* sporophytes (Lobban and Harrison 1994). On the other hand, the respiration rates for *L. hyperborea* sporophytes increased at only very high Zn concentrations of 250 mg/L (Hopkin and Kain 1978). Survival of germinating gametophytes of *L. hyperborea* was reduced in 0.1 mg/L Cu and 5 mg/L Zn, production of sporophytes from gametophytes was delayed 13 days by Cu at 0.025-.075 mg/L and by 2-4 days in Zn at 1000 mg/L. Growth was reduced by Cu at 0.001 mg/L and by Zn at 0.5 mg/L (Hopkins and Kain 1978). In a more recent paper, gametophytes of *L. japonica* could not survive in culture solutions containing more than 50 mg/L Cu and growth was restricted at 5 mg/ L (lowest concentration tested). Toxicity was more pronounced in mixtures of heavy metals (Ye *et al.* 2005).

Heavy metals from both natural and anthropogenic sources are common in marine waters, but they are not always bioavailable. Trace metals concentrations in sediments are largely dependent upon sediment grain size because they are associated with particle surfaces (Rae 1997; Loring 1991). Finer grained sediments, particularly within the clay and silt fractions, are comprised of metal-bearing minerals which may bind with, among others, Cu and Zn. These metals also strongly bind with particulate organic matter like that originating from fish farms (Chapman *et al.* 1998). To be bioavailable, metals must be in the dissolved form as free metal ions. The mobilization of sediment and organic particulate-bound metals into dissolved phase is controlled by oxidation. When particulate organic matter is oxidized (as happens when organically impacted benthos are left to recover between fish farming production cycles), organically-bound metals are released in soluble (bioavailable) form. The effects of metal bioavailability in sediments due to oxygenizing conditions were examined in a study conducted by Trannum *et al.* (2004). In that study, sediments containing high levels of Cu (400–1500 mg/kg) negatively effected colonisation by several taxa when this sediment was removed and positioned at a relatively pristine (well oxygenated) location in an outer fjord.

Hydrogen sulphide is commonly produced in anaerobic sediments where the bacterial community reduces sulphate in the absence of free oxygen, and such anoxic conditions are typical in the benthos below salmon farms after the waste matter has accumulated (for example, Hargrave *et al.* 1997; Johannessen *et al.* 1994). Sulphide is toxic to marine plants at low levels between 0.25-2.5 mM (Raven and Scrimgeour 1997). Toxicity due to sulphide (0.5 mM) had overriding negative impacts on survivorship and growth of the brown seaweed *Fucus* embryos buried by sediment in still water (Chapman and Fletcher 2002).

Hydrogen sulphide and the hydrosulphide anion are the most commonly found sulphide species in aqueous form, and hydrogen sulfide is the most apt to freely cross the membranes of aquatic organisms (Wang and Chapman 1999). Hydrogen sulphide, at a pH typical of seawater (8), forms approximately 9% of the total free sulphides (Erickson *et al.* 2001). Free sulphides can complex with heavy metals such as cadmium, copper and zinc in the sediment pore water, converting the sulphides in the free form into a non-toxic complexed form, and potentially reducing the availability of toxic free metal ions in the sediment pore water (Rittmann and McCarty 2001; Wang and Chapman 1999). Slight increases in sediment redox potential due to oxygenation does, however, cause metals complexed with sulfides to become mobilized and released in dissolved form into the pore spaces of the sediments.

In summary, sulphide generated from decomposition of organic matter affects seaweeds at very low concentrations. The heavy metals associated with sediments below and surrounding fish farms (for example, Zn and Cu) are not likely to be toxic to seaweeds under reducing conditions, but if these sediments become oxygenated, seaweeds within the zone of influence of the farm may be negatively affected by heavy metal toxicity.

Two broad categories of hazards exist, namely dissolved nutrients and particulate matter. From the above analysis, different exposure pathways exist. Depending on the length of the nutrient pulse, epiphytic algae on macrophytes could increase while the growth of the host macrophyte could decrease. The effect of increased levels of epiphytic algae on sori production is unknown, but the effect can be linked to the growth rate of the sporophyte (for example, the sori is directly proportional to the size of the lamina). One could easily assume that if at least 75% of the lamina was covered with epiphytic algae, then sori production would be greatly impaired. Dissolved nutrients can also directly impact kelp tissues by changing how the tissues store nutrients for growth at a later date and by increasing biomass to levels causing light limitation and therefore lower growth rates. A carbon to nitrogen ratio exceeding 20 is generally considered to be nitrogen limiting, so kelp kept at that level during the growing season would eventually die. No one knows what the biomass level is that leads to death, however, doubling the kelp biomass has shown to decrease the growth rate by 50%. This decrease in growth rate could be considered severe in the areas where kelp are under attack by sea urchins. Particulate matter from a fish farm has the potential to bury gametophytes, and increase the likelihoods of exposing kelp to toxic levels of sulphide and heavy metals. Gametophytes buried by 3 mm of sediment, and exposed to 0.25-2.5 mM of sulphide or 50 mg/L Cu die.

No field evidence exists to show that fish farming effluents affect kelp populations; however, there is solid theoretical and experimental evidence to show that there can be adverse consequences to individual macrophytes. Biological and past harvesting information indicates that an affected population would recover if the kelp bed extends over 200 m (for example, the spore dispersal range) from a kelp affected area (Some farm sites are very large, 300 m and longer) and/or only the older individuals are affected (so as to permit younger lower story individuals the opportunity to grow).

6.4.3 Risk Assessment

In this section a systematic process for describing and quantifying the risks associated with fish farming effluents at the two salmon farming sites on kelp is presented.

6.4.3.1 Release Assessment

Waste release from a fish farm depends on the fish number, feeding rate, feed conversion ratio, harvesting rate and fish size. The major factors are fish size, number and feed conversion as they dictate the total digested feed ration at a similar water temperature and feed type. Fish biomass grows nearly exponentially, but due to a decrease in growth rate with age and a changing feed conversion ratio, the amount of solid waste output grows nearly linearly (Figure 6.4.6, Buryniuk et al. 2006). Dissolved nitrogen (mostly ammonium) and phosphorus outputs follow similar trends.

A typical fish production cycle is 60 weeks (faster, or slower depending on the water temperature). Young Atlantic salmon (*Salmo salar*) enter the sites between January and May, and this means the largest biomass (and waste output) would occur before harvesting start dates 14 to 16 months later (April to August). Fish are not harvested all at once, but staggered over an extended period of time. Another production cycle can begin right after all fish have been harvested.

6.4.3.1.1 Zone of dissolved nutrients

The total loading of N and P from a typical salmon farm is 78 and 9.5 kg per ton of fish produced, and 57-86% and 22-46% is ammonium and dissolved P, respectively (Ackefors and Enell 1990, Fivelstad *et al.* 1990). Norwegian aquaculture regulations do not specify that nutrients must be monitored on fish farms (Hansen et al 2001); therefore, empirical data were used to estimate the magnitude and zone of dissolved nutrients that surrounds a fish farm. As nitrogen is normally limiting in temperate coastal waters, we assumed that dissolved N is the only limiting nutrient, and limits kelp production outside of the springtime period. Estimates generated from existing empirical models could have given us daily outputs of dissolved nitrogen, but as they do not consider dispersion and other dilution effects due to wave and current, and cage structure, we could not use them to estimate nutrient levels at the kelp beds. Furthermore, existing models do not consider fluctuations during feeding time in ammonium levels; a variable that plays an important role in algal epiphyte development. The empirical data used to estimate the zone of ammonium are described next.

In a relatively low wave exposed farm site containing approximately 3 kg sized fish with currents averaging 0.024 m/s over a 24 h period, Ahn *et al.* (1998) found that ammonium levels fluctuated during summer between fish feeding and non-feeding times to often exceed 10 µM, while background nitrate levels remained fairly constant at approximately 2.5 µM (Figure 6.4.7). Increases in ammonium near fish cages associated with feeding events were also observed in Chile and elsewhere in Canada (Soto and Norambuena 2003; Wildish *et al.* 1993). Soto and Norambuena (2003) suggested that surrounding the farm, dilution and nutrients recycling must be high, as there was no connection between nutrient conditions in the water column and the levels of N and P in the sediments due to organic waste loading. In the Bay of Fundy, Canada, where flushing conditions are strong, levels of nutrients outside of fish farms during the summer months were indistinguishable from the nutrient poor background conditions (Wildish *et al.* 1993). The authors of that study attributed this to nutrient uptake by microalgae and high flushing conditions.

Available dissolved nutrients from the salmon farm suffer dilution due to dispersion and wave effects. In Petrell *et al.* (1993) existing empirical data on dissolved nitrogen was used to describe the change in its concentration up to 40 m from a salmon farm. The available data appear to indicate that ammonium levels are lowest within the first 10 m of the sea cage and tend to randomly fluctuate up and down within an apparent mixing zone found between 10 to 40 m from the farm (data :Black and Carswell 1987; Korman 1989; Weston 1986). Ammonium between 25 to 40 m from the cage varied from 2.3 to 5.6 µM, and most likely these values depended on the fish biomass and current conditions at the time of sampling. Ammonium levels at 40 m are expected to be higher today because at the time of those investigations, on-farm biomass and farm sites were much lower as compared to today’s standards.