Fisheries and Oceans Canada
Science

Pêches et Océans Canada
Sciences

C S A S

Canadian Science Advisory Secretariat

S C C S

Secrétariat canadien de consultation scientifique

Research Document 2005/034

Document de recherche 2005/034

Not to be cited without
Permission of the authors *

Ne pas citer sans
autorisation des auteurs *

Eutrophication Impacts of Marine Finfish Aquaculture

Répercussions de l'eutrophisation due à l'élevage de poissons marins

Peter M. Strain

Ecosystem Research Division
Bedford Institute of Oceanography
PO Box 1006
Dartmouth, NS B2Y 4A2

* This series documents the scientific basis for the evaluation of fisheries resources in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

* La présente série documente les bases scientifiques des évaluations des ressources halieutiques du Canada. Elle traite des problèmes courants selon les échéanciers dictés. Les documents qu'elle contient ne doivent pas être considérés comme des énoncés définitifs sur les sujets traités, mais plutôt comme des rapports d'étape sur les études en cours.

Research documents are produced in the official language in which they are provided to the Secretariat.

Les documents de recherche sont publiés dans la langue officielle utilisée dans le manuscrit envoyé au Secrétariat.

This document is available on the Internet at:

Ce document est disponible sur l'Internet à:

http://www.dfo-mpo.gc.ca/csas/

ISSN 1499-3848 (Printed / Imprimé)

Canada

ABSTRACT

This paper describes eutrophication and examines how wastes from netpen finfish aquaculture contribute to that process. A number of eutrophication definitions are examined, with the conclusion that eutrophication is essentially an increase in the flow of energy and cycling of nutrients by the ecosystem. Two eutrophication classification schemes and their associated indicators are discussed, showing that eutrophication must be considered in the context of individual environments. Mass balance models that are used to calculate the amounts of eutrophying wastes from finfish farms are described, together with other measurements that yield information about the fates of aquaculture wastes. Results of these calculations show that most dissolved and particulate wastes from farm sites are transported far from farms, even ones located in depositional environments. Given the right combination of the intensity of farming and the carrying capacity of the receiving environment, finfish aquaculture can produce eutrophication impacts on scales of kilometers to tens of kilometers and can change the structure and functioning of the ecosystem in significant ways on these scales. Finally, the management implications of eutrophication caused by finfish aquaculture are discussed with a focus on those processes that influence environmental quality on large scales, the so-called far-field or bay-wide effects. A management approach that does not consider the potential for large scale eutrophication may be ignoring one of the most significant potential impacts of some marine finfish culture operations.

RÉSUMÉ

Ce document décrit le processus d'eutrophisation et examine comment les déchets d'élevage de poissons en enclos contribuent à ce processus. L'examen d'un certain nombre de définitions d'eutrophisation a permis de conclure qu'il s'agit essentiellement d'une augmentation des transferts d'énergie et du cycle des éléments nutritifs au sein de l'écosystème. Une discussion sur deux systèmes de classification de l'eutrophisation et leurs indicateurs connexes a permis d'établir que l'eutrophisation doit être prise en considération dans le contexte des milieux individuels. Les modèles de bilan massique utilisés pour calculer les quantités de déchets eutrophisants produits par les établissements d'élevage de poissons sont décrits, tout comme d'autres mesures qui fournissent des données sur le devenir des déchets aquacoles. Les résultats de ces calculs montrent que la plupart des déchets dissous et sous forme de particules des établissements d'élevage sont transportés sur de grandes distances, même dans le cas des établissements situés dans des milieux de dépôt. En présence d'une bonne combinaison d'intensité d'élevage et de capacité de charge du milieu récepteur, la pisciculture peut entraîner une eutrophisation du milieu sur des kilomètres ou des dizaines de kilomètres et peut modifier considérablement la structure et le fonctionnement de l'écosystème à ces échelles. Finalement, les conséquences de l'eutrophisation due à la pisciculture sur la gestion sont abordées en mettant l'accent sur les processus qui influent sur la qualité du milieu à grande échelle, c.-à-d. les effets à distance. Une approche de gestion qui ne tient pas compte du risque d'eutrophisation à grande échelle ne prend peut-être pas en considération une des répercussions possibles les plus importantes de certaines activités d'élevage de poissons marins.

INTRODUCTION

Sea-cage salmon aquaculture discharges a number of waste types into the marine environment. Included in these wastes are dissolved inorganic nutrients (nitrogen and phosphorus) and particulate organic matter. The organic wastes include waste feed and salmon faeces that are partially decomposed by bacteria. This decomposition process consumes oxygen, adds additional nutrients and lowers the dissolved oxygen content in both the water column and surface sediments. The respiration of the farmed fish further reduces the dissolved oxygen levels. In areas of intensive aquaculture, these waste streams can have significant impacts on the ecosystem: organic wastes may smother or otherwise alter benthic habitat, nutrients can stimulate growth of phytoplankton and/or attached algae, and reduced levels of dissolved oxygen can stress native organisms as well as the farmed fish. The build-up of nutrients, the reduction of dissolved oxygen levels and the stimulation of algal growth are all aspects of a process known as eutrophication. Eutrophication impacts occur on both local and bay-wide scales, and may be a significant concern for the environmental sustainability of aquaculture in areas of high density farming.

This paper will define and describe eutrophication and will examine how wastes from netpen finfish aquaculture contribute to that process. Mass balance models that are used to calculate the amounts of wastes from farm operations will be discussed, and the fractionation of farm wastes into dissolved and particulate wastes will be quantified. The significance of these waste streams will be compared to both natural processes and wastes from other human activities. The quality of these estimates of waste discharges will be discussed throughout. Finally, the management implications of eutrophication caused by finfish aquaculture will be discussed with a focus on those processes that influence environmental quality on large scales, the so-called far-field or bay-wide effects. In much of what follows, the salmon aquaculture industry in southwestern New Brunswick (SWNB) will serve as a case study, but the focus will be on deriving principles that are general.

WHAT IS EUTROPHICATION?

Many different definitions of eutrophication have been used: Nixon (1995) noted that approximately a dozen different definitions were identified by a series of workshops on estuarine eutrophication in the United States. In an attempt to simplify this confusion, he proposed the following definition:

> *eutrophication*: *an increase in the rate of supply of organic matter to an ecosystem.*

In proposing this definition, Nixon respected the historical use of the concept of eutrophy (or nourishment) in medicine and recognized some important subtleties in the description of eutrophication. Nixon explained that his definition includes both autochthonous organic matter (i.e. that which is produced *in situ*) and allochthonous organic matter (i.e. that which is imported from elsewhere), and may be the result of increased inputs of inorganic nutrients, which lead to higher *in situ* productivity rates, or direct inputs of organic matter. He used the term to describe an increase in the amount of energy available for the metabolism of an aquatic ecosystem over time rather than a description of its current status.

Descriptions of trophic status, which may be thought of as indicators of overall productivity, have a long use in marine ecology and have spawned the terms: oligotrophic, mesotrophic, eutrophic and hypereutrophic, going from least productive (or energy poor) to most productive (energy rich). On this basis, an ecosystem may be eutrophic but not undergoing eutrophication, if it is in a stable state. Sometimes such ecosystems are called eutrophied or eutrophicated to indicate that eutrophication has occurred in the past. The second distinction that Nixon made was that eutrophication is a change over time in the functioning of the ecosystem, rather than the causes of such a change. He argued that increases in inorganic nutrients that do not produce a change in productivity are not eutrophication. On this point his definition is not compatible with the following one from Schramm and Nienhuis (1996):

the process of natural or man-made enrichment with inorganic nutrient elements.

However, enrichment with nutrients has the potential to increase productivity, even if such increases are not closely tied to the source of the nutrients. By including natural processes in their definition, Schramm and Nienhuis considered spatial differences in inorganic nutrients to be examples of eutrophication as well as temporal differences. Other eutrophication definitions describe the whole process in more detail, such as this one from the Oslo and Paris Commission (OSPAR, 1998):

Eutrophication means the enrichment of water by nutrients causing an accelerated growth of algae and higher form of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned, and therefore refers to the undesirable effects resulting from anthropogenic enrichment by nutrients.

In this case the nature of the nutrients is not specified, and one presumes that they may be inorganic or organic forms. The semantic distinctions aside, it is important to recognize that eutrophication is about the intensity of energy flow and nutrient cycling in an ecosystem, and that the symptoms of eutrophication are many and varied. This last point is explicitly recognized in a definition of eutrophication from a website of the Danish Environmental Protection Agency, which includes descriptions of the primary and secondary effects in the definition of eutrophication:

…the primary or direct effects include: increased primary production, elevated levels of biomass and chlorophyll a concentrations, shift in species composition of phytoplankton, and shift from long lived macroalgae to short lived nuisance species.

The secondary or indirect effects include increased or lowered oxygen concentrations, and changes in species composition and biomass of zoobenthos. … eutrophication can result in impoverished biological communities and impaired conditions.

Finfish aquaculture discharges clearly meet all of these definitions of eutrophication, as they include both inorganic nutrients and organic matter and are a change in nutrient inputs in time and space. Moreover, the demand of fish respiration for dissolved oxygen directly impacts the concentration of dissolved oxygen, which is usually considered a symptom of eutrophication rather than a cause. All of these influences of aquaculture can potentially change the functioning of the ecosystem in ways that are embodied by the concept of eutrophication (Fig. 1). In temperate waters, the potential for eutrophication impacts from aquaculture will be highest in the late summer and fall when fish biomass and metabolic processes are high, and water temperatures are at maximum and dissolved oxygen at the minimum of their seasonal cycles. Local impacts may include organic matter enrichment of the benthos, with resulting changes in the benthic community, or in extreme cases smothering of the benthic community; depressed water column O_2 levels that stress the farmed fish; and elevated nutrient concentrations that may stimulate increases in productivity. Far-field impacts may include higher levels of organic matter available for ecosystem metabolism, either directly or indirectly through increases in productivity driven by inorganic nutrients; depressed O_2 levels that stress wild organisms; or even changes to the overall structure and functioning of the ecosystem. Eutrophication related discharges from aquaculture can be discussed in terms of organic carbon or organic matter loadings, nutrient loadings or oxygen demand. Because nitrogen is usually the limiting nutrient in coastal marine waters, the literature has generally focussed on organic and inorganic forms of nitrogen.

The severity of eutrophication symptoms will always be a balance between the magnitude of nutrient inputs and the capacity of the ecosystem to absorb those stresses. If nutrient inputs from aquaculture are small compared to natural or other anthropogenic inputs, the additional disturbance to the ecosystem is likely to be small. Exactly how nutrient inputs are manifested can also be important. For example, in the salmon growing areas of southwestern New Brunswick (SWNB), the gross fluxes of nutrients exchanged with offshore waters by the tides, measured in mass per unit time, are very large compared to any anthropogenic inputs. However, the concentrations of nutrients in the inflowing tides are not very different from natural background levels. As will be shown below, however, aquaculture discharges in late summer and fall can significantly increase ambient nutrient levels. Concentrations in excess of half saturation constants for growth of phytoplankton or macroalgae are much more likely to have direct impacts on productivity than the very low levels in waters from offshore. The capacity of the ecosystem to absorb excess nutrients or oxygen demands is largely a function of the setting (e.g. is it an inlet with restricted mixing with offshore, an inlet with good exchange with the offshore, an open coastal environment?) and circulation patterns. It also depends on the natural metabolism of the ecosystem: how large a change in ecosystem functioning might be caused by additional nutrient inputs? All of these considerations are very scale dependent, and an assessment of how the various factors change as the spatial scale increases away from the nutrient source is critical to understanding the potential for eutrophication impacts.

MEASURES OF THE SEVERITY OF EUTROPHICATION

Because eutrophication of coastal marine waters is a significant global issue, a great deal of effort has gone in to researching both the eutrophication process and management tools for classifying the potential for eutrophication and eutrophication impacts. The U.S. National Oceanic and Atmospheric Administration (NOAA) conducted a national classification of eutrophication conditions in U.S. estuaries using a scheme based on three "primary symptoms" (chlorophyll *a* concentrations, problematic epiphytic growth, and problematic macroalgal growth) and "three secondary symptoms" (low dissolved oxygen levels, submerged aquatic vegetation (SAV) losses, and nuisance and toxic blooms of algae; Bricker at al. 1999). While these indicators appeared satisfactory for the purposes this scheme was designed for, it is instructive to examine each of these indicators in the context of conditions in SWNB. The tidal flows in SWNB cause inshore waters to be highly turbid, so that water column productivity is probably limited by light, rather than nutrient supply (Harrison et al., 2005). Since productivity is restricted by turbidity, increases in water column chlorophyll *a* or the underlying benthic epiphytes are unlikely. Macroalgal growth in the intertidal zone has been observed, and tracers show that waste discharges from fish farms are reaching the intertidal zones more than a kilometer away from the farms (Robinson et al., 2005). Strain and Hargrave (2005) also noted that oxygen consumption rates in intertidal sediments in salmon farming areas in SWNB were higher than those measured on the seabed adjacent to cages or in intertidal sediments far removed from aquaculture areas, suggesting that impacts of aquaculture may be focussed in the intertidal zone in SWNB. Based on the NOAA classification scheme, macroalgal growth is therefore the only potentially useful primary symptom in SWNB. But the actual use of macroalgal growth as an indicator is complicated by a lack of historical data (which could be addressed through a new monitoring program to track macroalgal growth) and the difficulty of establishing a clear cause-effect relationship between aquaculture discharges and growth in a remote intertidal zone. The latter is a general comment: as the scale of eutrophication impacts increases, attributing cause becomes more difficult.

Similar comments can be made about the application of the NOAA secondary symptoms to SWNB: low dissolved oxygen levels are observed (eg. Page et al., 2005), but how to apply O_2 levels as an indicator is uncertain; SAV loss is probably not applicable (measures of eel grass extent are the most common indicators for SAV loss: eel grass beds are not common in SWNB); there is a long history of naturally occurring toxic blooms in SWNB, but the causes for their appearance are poorly understood and proof that the frequency of such blooms were or were not increasing due to aquaculture would not be possible at this time. The point here is not that the NOAA indicators are inappropriate measures of eutrophication, but that indicators for eutrophication must be chosen to suit the particular receiving environment in which they are used.

Other schemes for classifying eutrophication have also been developed. For example, the European Environmental Agency (EEA) has used a scheme that is based on winter nutrient levels, chlorophyll *a* and bottom water O_2 levels (e.g. EEA, 1999; EEB, 2001). Newton et al. (2003) compared the results of eutrophication classifications based on NOAA and EEA schemes for the Ria Formosa in Portugal. The EEA criteria suggested that conditions in this lagoon were poor; the NOAA criteria suggested the lagoon was nearly pristine. This contradiction is a further example of why eutrophication must be evaluated in the context of local conditions.

Given these observations, it is unlikely that any single eutrophication classification scheme will meet the needs for management of aquaculture in Canada. A more practical alternative is to assess the impacts of aquaculture activities by making the following types of comparisons within the local context:

- How do the inputs of nutrients/demand for oxygen from aquaculture compare with natural and other anthropogenic inputs?
- To what extent can aquaculture discharges influence ambient concentrations of nutrients and oxygen?
- How does the processing of nutrients / energy by aquaculture compare with the natural ecosystem metabolism?

Finally, assessing eutrophication impacts of aquaculture must be done in the context of the overall management goals. Eutrophication by itself is judgement free: some eutrophication of a low productivity area may be seen as beneficial; a small amount of additional eutrophication may be seen as detrimental in an environment that is already eutrophic. Recent trends to think of ecosystem-based management goals are useful here: setting a goal for an ecosystem that has minimal disturbance from a natural state will result in very different management actions than setting a goal that is directed solely at the sustainability of the aquaculture activity. Without a conscious decision of what we are managing for, the selection of management tools will remain elusive.

A CASE STUDY: SOUTHWESTERN NEW BRUNSWICK

The salmon industry in southwestern New Brunswick (SWNB) will be used to illustrate the process described above for the evaluation of potential eutrophication impacts from finfish cages. This section will describe the sequence of calculations necessary to assess the amounts of the different wastes discharged, the fates of those wastes, and their environmental significance. The discussion will describe both the general requirements for information at each step in the process and how those requirements were met with specific data available for SWNB. More details of the SWNB calculations are provided in the Appendix and in Strain and Hargrave (2005).

Waste Discharges from Aquaculture

In land-based aquaculture, it is feasible to measure waste outputs directly by monitoring the composition of the waste stream. This is not possible in marine sea-cage aquaculture. The alternative is to use a mass balance approach to compare the inputs to the farms (salmon smolts, feed) with the outputs (mortalities, escapes, and harvested fish). By a consideration of the elemental or proximate composition of the feed and fish and some knowledge of salmon metabolism, it is possible to estimate nutrient discharges from and oxygen consumption due to the fish based on the differences between inputs and outputs. To achieve adequate temporal resolution, the data on inputs and outputs must be available on a monthly basis. Such detailed information on farm operations is available in some jurisdictions (e.g. Norway: Schaaning and Kupka Hansen, 2005), but is not always available in Canada and may be considered proprietary. There are many steps to such mass balance calculations of aquaculture wastes, and different workers have used different combinations of data, models, and assumptions in making these estimates.

Fig. 2 shows the steps taken to calculate dissolved and particulate waste discharges from salmon cages in SWNB. The first requirement is for information on the size and growth of fish throughout the grow-out cycle. Such information could come directly from industry data or, as in the SWNB case, the seasonal temperature cycle and a fish growth model can be used to predict the size and growth rate of a single salmon (Fig. 3). In step 2 (Fig. 2), information on the amount of feed used is required so that the waste can be calculated from the difference between feed used and fish growth. Such information could come directly from industry feed logs, or feed use can be calculated from feed conversion ratios (FCRs). In step 3, data on the amount of C, N and P in feed and fish are required to determine how much of each is added in the feed, and how much is incorporated in the fish tissues. The composition of feed is routinely provided by feed suppliers, although it may be necessary to convert proximate composition data to elemental compositions (see Appendix). Extensive data on the proximate and elemental composition of fish are available in the scientific literature and can be used if direct measurements are not available from the local industry. In step 4, data on salmon nutrition (apparent digestibility coefficients, carbon retention coefficients etc.) are used to estimate the initial partitioning between dissolved inorganic wastes and particulate organic wastes. Fig. 4 shows how the carbon, nitrogen and phosphorus in feed is initially partitioned into growth (termed retained C, N, or P in the fish nutrition literature), metabolic consumption (respiration for carbon, urine for nitrogen and phosphorus), particulate faeces and waste feed as fish are fed on farm sites. In step 5 (Fig. 2), these fractions are adjusted to account for the rapid decomposition of some of the solid wastes (faeces and waste feed). Some fraction of the solid wastes is very labile to decomposition by the microbial community. Strain and Hargrave (2005) reported measurements of O_2 uptake rates over a period of 20 days by particles that were resuspended from heavily biofouled nets, and in surface sediments containing high concentrations of feed pellets and faeces and estimated that ~50 % of the available carbon in the particulate farm wastes were oxidized within 5 days. The net effect of this decomposition is to convert particulate N and P to dissolved N and P and to create a biological oxygen demand, thus reducing the solid wastes that will accumulate in sediments and increasing the dissolved wastes that will affect water column conditions locally and disperse more widely. It is important to choose a time scale for this estimate that is appropriate to the receiving environment: 5 days is appropriate for the salmon farming areas in SWNB because the residence times of water in these areas are of this order. After adjusting for this decomposition, it is possible to predict the fractions of the wastes in the dissolved form and the refractive portion of the particulate wastes that are likely to be permanently buried in the sediments (Fig. 5). These calculations do not indicate whether material buried in sediments will be close to or far from farm sites.

So far this sequence of calculations has predicted the solid and dissolved wastes from individual fish. In step 6 (Fig. 2), waste production on a farm scale is calculated using the number of fish on site to predict near-field impacts. The numbers of fish could be determined from actual farm records or from management controls such as approved production limits (APLs). The total discharges of carbon, nitrogen and phosphorus, and the total oxygen demands, are shown in Table 1 as kilogram of waste per tonne of fish produced. The waste estimates may be expressed in many ways: e.g. Strain and Hargrave

(2005) also reported wastes as the maximum daily discharge per 1000 fish during each calendar year of the grow-out cycle (their Table 2) and noted that representing discharges in these terms facilitated the updating of impact estimates as farm locations and stocking levels change over time. Finally in step 7 (Fig. 2), the total impacts of many farm sites can be calculated by a consideration of the distribution of fish at each farm and their breakdown by year class, variations in grow-out cycles etc. Strain and Hargrave (2005) also reported waste estimates for each 'Coastal Management Region' in SWNB (their Table 3).

As can be seen from the description above, estimating eutrophication wastes from salmon aquaculture involves a lengthy series of calculations, with many different input parameters. How reliable are these estimates? It is possible to determine the uncertainties in some steps of these calculations in a reasonably rigorous way (e.g. Strain and Hargrave, 2005), but error assessment of the whole sequence of models / calculations requires a statistical approach. The Appendix describes a Monte Carlo approach to estimating the precision of the model predictions from estimates of the uncertainties in the input parameters.

For model outputs expressed per tonne of fish produced, uncertainties in the predicted dissolved wastes and O_2 demand (i.e. the water column impacts) range from 13 – 15 %; uncertainties in predicted particulate wastes (i.e. the sediment impacts) range from 24 – 52 % (Table A.2). The environmentally significant carbon impact is the particulate fraction; its precision is estimated at 34 %. Both the dissolved and particulate nitrogen fractions are potentially important, but dissolved nitrogen is more important for far-field eutrophication impacts. It accounts for ~78 % of the total nitrogen waste, and has a precision of 15 %. The biggest contributors to the uncertainties in wastes expressed per tonne of fish produced are the metabolic (respiration or urine) / retained ratios, the FCR, and the composition of feed. None of the parameters that go into the growth model (TGC, Initial smolt weight etc) have much impact, because the normalization to fish produced removes such influences.

For model outputs expressed as the maximum discharges per day in the second calendar year of the grow-out cycle, the uncertainties range from 22 – 56 % (Table A.3) and are all higher than they were for the wastes per tonne of production. This is because the maximum one-day discharge depends on fish size which in turn depends on water temperature, and is influenced most strongly by uncertainties in the TGC value. However, the other patterns for wastes expressed this way are similar to those for the per tonne production, with precisions of dissolved wastes (22 – 23 %) better than the precisions of particulate wastes (31 – 56 %), and the precision of dissolved N about twice as good as that for particulate C.

Another way of evaluating the precision of these calculations is to compare them with estimates for modern farm operations in other areas. For example, a conceptually similar mass balance approach is used for the management and regulation of aquaculture operations in Scotland (Gillibrand et al., 2002). Even though the dissolved and solid nitrogen estimates based on their approach are for different growing conditions in a different environment using a different set of available data and assumptions, the agreement between the waste nitrogen release estimates for SWNB and Scotland is good (a difference of ~15 %, Table 1). Gillibrand et al. (2002) also present correction factors to convert the waste estimates for Atlantic salmon to values for other species. As with the calculations described here for SWNB, estimates of fish numbers and size, or biomass, on a site by site basis and some means of estimating feed use are essential for these calculations. Other estimates of the wastes from finfish aquaculture drawn from the literature are also given in Table 1 for comparison. Earlier estimates tended to predict greater nitrogen losses, and may be a reflection of lower feed efficiencies in the past. When making these estimates, it is important to use up-to-date information on feed use that reflect current industry practices.

Models like DEPOMOD combine estimates of the discharge of faeces and waste feed with current patterns and bathymetry to predict deposition patterns around finfish farms, and are increasingly used for management purposes (e.g. Chamberlain et al., 2005). The results of such models can be compared with the mass balance approach described here. It is possible to combine the predicted fluxes of faeces and waste feed (prior to the remineralization of the wastes) from the mass balance models with estimates of the fraction of carbon (~0.5) in and the dry density (~0.6) of dry faeces and waste feed and information of the density of fish at a site to predict local deposition and sedimentation rates that would result from the delivery of all faeces and waste feed to the sediments. For example, Table 2 contains the mass balance predicted deposition and sedimentation rates for year 2 of the grow-out cycle for a site in SWNB with 80,000 fish in 9 polar circle cages for a range of spreading factors (i.e. the ratio of total cage area to sediment area impacted). Deposition rates are given as daily rates on the day of maximum discharge (Oct 2); sedimentation rates are averaged over the entire calendar year.

Fates of Aquaculture Wastes

There is an extensive literature on the accumulation of solid wastes below marine finfish sites, and the effects that these deposits have on the benthic environment immediately below and close to the cages. These impacts are clearly a result of eutrophication: such local benthic effects are described in other background papers for this meeting. But in assessing the relative importance of local and far-field impacts, it is necessary to know what fraction of the wastes accumulate in the debris piles under cages and which are transported further from the farms. Strain and Hargrave (2005) reported on two different calculations that were used to assess the percent of wastes that accumulate on the seabed immediately adjacent to the cages. One was based on a comparison between waste fluxes from the cages and O_2 uptake and nutrient release measurements from both fouling communities and sediments that contained large amounts of feed pellets and faeces near cages; the second compared waste discharges with sedimentation rates based on dated sediment cores. Both of the approaches suggested that only ~ 1 % of the total solids discharged (~2 % of the refractory solids) from farms were found in sediments close to farm sites in Lime Kiln Bay, a highly depositional environment in SWNB (Fig. 6). These estimates are not well constrained, but strongly suggest that most of the solid wastes from finfish sites do not accumulate near the cages, even at depositional sites. Another uncertainty over the fate of farm wastes is due to the fouling communities that accumulate on net pens and other gear at salmon farms. In SWNB in recent years, net cleaning and disposal of the resulting wastes have been on shore and therefore should not be included in estimates of the marine impacts of aquaculture. Hargrave (2003) reported fouling communities on nets consisting mostly of mussels (*Mytilus edulis*) with a wet weight of 3.5 kg m^{-2}. Other observations report a mussel density of 2.2 kg m^{-2} (Shawn Robinson, St. Andrews Biological Station, personal communication), and note that other organisms were present as well. Despite the limited data availability, it is possible to use this information to estimate the nitrogen trapped in the fouling community by making a number of assumptions:

- total fouling community equivalent to 4 kg m^{-2} of mussels, uniformly distributed over the entire area of the sides of the net.
- fouling densities reach this level seven times during the grow-out cycle
- typical net pen geometry in SWNB (circular net with a circumference of 100 m and a height of 7 m)
- fish production in cage equivalent to the recommended stocking density (18 kg m^{-3}).

Using these assumptions, which are thought to err towards overestimating the fouling community on the nets, the nitrogen trapped in the fouling community represents 3.2 % of the total nitrogen waste from the cage. The overall fate of the nitrogen from farm wastes is summarized in Fig. 6. The calculations for both the accumulation of nitrogen in sediments near cages and the amount of nitrogen trapped in the fouling communities are both very approximate and could both be improved if better data were available. However, both calculations suggest strongly that these terms are not large contributors to the nitrogen budget. Revisiting these calculations is very unlikely to change the general nature of the principal conclusions to be drawn from this analysis: most nitrogen waste is in dissolved form and most of the nitrogen waste in particulate form is transported away from a farm. Both of these conclusions imply that far-field eutrophication impacts must be considered in assessing potential environmental impacts of finfish aquaculture.

Assessing the Environmental Significance of Aquaculture Wastes

Discharges from finfish aquaculture can be put into environmental context in a number of ways. Strain and Hargrave (2005) evaluated aquaculture wastes in the salmon farming region of SWNB (Fig. 7) by comparing aquaculture discharges with other anthropogenic and natural fluxes, by assessing the changes in ambient nutrient concentrations that could result from aquaculture discharges, and by comparing the cycling of carbon, nitrogen and oxygen by salmon growing operations with that of the natural ecosystem.

In SWNB, the Letang Inlet (Fig. 8) receives wastes from a pulp mill, a large fish processing plant, and a municipal sewage treatment plant, and is the salmon farming area most impacted by other human activities. Figure 9 compares salmon aquaculture discharges with those from these other sources and from precipitation and runoff. Aquaculture discharges predicted for this area for odd years are similar to those predicted for the Letang Inlet in 1992 prior to introduction of single year class management in the area (Strain et al., 1995). However, discharges predicted for even years are approximately 2.4 times higher than they were in 1992. This switch to single year class management has exacerbated potential environmental impacts in those areas with the highest stocking density, because most smolts are introduced to this area in odd years. Hence mature fish occur at most sites during one year, instead of being spread out over two years. Marked increases in APLs in recent years have also contributed to this concern. In 1992, the aquaculture industry was clearly the largest anthropogenic contributor of oxygen demand and nutrients to the Letang Inlet. The dominance of salmon aquaculture sources has continued: in odd years salmon aquaculture discharges are now comparable to those from the second largest anthropogenic source (the fish plant); in even years, discharges from salmon aquaculture are estimated to be 1.6 to 3.5 times greater than the second largest anthropogenic source.

Strain and Hargrave (2005) combined information on the residence times of water in different 'Coastal Management Areas' (CMRs, Fig. 7) in SWNB with the aquaculture discharge estimates to calculate the changes that might occur in the ambient levels of nitrogen, phosphorus and dissolved oxygen (Table 3). These estimates are for the most sensitive time of year, because they are based on the maximum daily waste discharges, which occur in early October, and for the year of the odd/even cycle of year classes with the higher discharges. The predicted changes in ambient oxygen and nutrient concentrations for the Northern Passamaquoddy Bay and Deer/Campobello Island CMRs are very small. A decrease of 1.4 µM for dissolved oxygen is ~ 0.5 % of oxygen saturation (275 µM) at this time of year. Changes of 0.18 µM in nitrogen and 0.012 µM in phosphorus are much smaller than the concentrations of these nutrients in offshore waters in the Bay of Fundy. But the predicted changes for the Letang CMR are very significant. A 43 µM decrease in oxygen is equivalent to a decrease of 16 % in oxygen saturation or of 1.4 mg l^{-1} in oxygen concentration. An increase in dissolved inorganic nitrogen of 5.5 µM would more than double observed average background levels. Field data confirm that very high concentrations of nitrogen do occur in these waters. Bugden et al. (2001) reported inorganic nitrogen concentrations as high as 9.7 µM in Back Bay (see Fig. 8) in September 1999. The increased availability of inorganic nitrogen could be promoting increased biomass of macroalgae in adjacent intertidal areas. Thus intensive aquaculture as practiced in certain areas of SWNB is capable of changing ambient oxygen and nutrient concentrations over scales of many kilometers.

Strain and Hargrave (2005) also compared the processing of oxygen, nitrogen and carbon by salmon aquaculture with the processing of these elements by natural processes (Table 4). The numbers in Table 4 are the ratios of the total fluxes due to salmon aquaculture to the total fluxes from natural processes, expressed as percentages. The totals for aquaculture include: for O_2: the O_2 consumed by both salmon respiration and the decomposition of waste feed and faeces; for N: the N released in dissolved form, during the decomposition of waste feed and faeces, and the organic N buried in the sediments, but not the N in the salmon biomass, which is removed from the ecosystem at harvest time; for C: the C respired by salmon, the C respired during the decomposition of waste feed and faeces and the C buried in the sediments, but not the C in the salmon biomass. Natural oxygen fluxes include water column respiration (including macrophytes) and sediment oxygen consumption; natural nitrogen fluxes include ammonia regeneration (phytoplankton + macrophytes) and sediment regeneration; natural carbon fluxes include primary production by phytoplankton and macrophytes. The amount of O_2 consumed by salmon respiration and the breakdown of wastes from salmon farms varies from less than 0.1 (northern Passamaquoddy Bay) to 20 % (Lime Kiln Bay) of the natural ecosystem metabolism, a range which probably corresponds to minimal to significant impacts on the ecosystem. The extra O_2 demand from farms in three of the four sub-regions of the Letang Inlet CMR are >9 % of the natural O_2 cycle. Salmon respiration in the semi-enclosed Cobscook Bay is <1 % of natural respiration, as are the values in each of the three other CMRs outside of the Letang Inlet.

The division of the Letang Inlet CMR into sub-regions (Fig. 8) clearly shows the importance of considering scales when doing this kind of analysis. O_2 consumption due to salmon farms is 2.6 % of the natural metabolism in the Letang Inlet as a whole, but 20 % in Lime Kiln Bay within that CMR: local impacts in smaller sub-regions can be much greater than in the CMR as a whole. Impacts at individual farm sites will be greater still. Such calculations show why conditions of sub-optimal dissolved oxygen concentrations requiring re-aeration in late summer months have occurred at some farms with restricted circulation and relatively long water residence times in the Letang Inlet CMR.

On the basis of the comparisons in Table 4, the cycling of nitrogen is more perturbed by salmon aquaculture than oxygen or carbon. In Lime Kiln Bay, nitrogen flux attributable to salmon in even years is 3.3 times higher than natural nitrogen fluxes. This means that more nitrogen is introduced to the ecosystem through salmon farming than is cycled naturally in the water column and sediments. Cultured fish are now a major biogeochemical pathway for nitrogen in both the bays that make up the Letang Inlet CMR and some of the larger CMRs. As stated above, these nutrient fluxes are evident in unusually high dissolved inorganic nitrogen concentrations, and may be responsible for promoting growth of intertidal macrophytes and other algae.

O_2 demand and nitrogen release discussed above can be considered net fluxes attributable to aquaculture (the nitrogen in the salmon biomass is not included in these estimates, since it is removed from the ecosystem during harvest). The processes either use resources from the ecosystem or may induce changes in its functioning. The situation for carbon is slightly different. Since the impact of excess CO_2 from salmon respiration is likely to be small in seawater that has high natural levels of dissolved CO_2, the carbon that is respired has little impact on the receiving environment: its addition does not pose a threat, and its production does not rely on wild biota. The carbon processed by fish respiration accounts for ~49 % of the carbon in aquaculture fluxes (Table 4). In Lime Kiln Bay, 164 % as much carbon is cycled through aquaculture than is processed naturally by the ecosystem if the carbon respired by the salmon is included. But even if the respired carbon is not included, the fluxes of carbon due to aquaculture are still more than 80 % of those due to natural processes. Salmon aquaculture also plays a very significant role in carbon cycling in Bliss Harbour and Back Bay, but a relatively minor role in Letete Passage and the other CMRs.

These comparisons between salmon aquaculture and the natural ecosystem metabolism are perhaps the most direct indicator of eutrophication, as they measure the extent to which salmon aquaculture alters both the structure and the functioning of the ecosystem. In cases where elemental processing by aquaculture significantly changes the total processing of these elements, clearly the functioning of the ecosystem has been altered. In order to do that, the salmon in the farms must be considered a new dominant species, so that the structure of the ecosystem has been altered as well. These conclusions are once again dependent on scale: local impacts will be more intense than those over larger scales. Similar comparisons between land-based agriculture and the natural terrestrial ecosystem would lead to a similar a set of conclusions which would also be scale dependent. Perhaps the most significant difference between these two types of food production however, is the difficulty of accurately describing the intensity and spatial distribution of impacts from marine finfish aquaculture.

MANAGEMENT IMPLICATIONS

To date, the primary focus of the management of marine finfish aquaculture in Canada has been on benthic impacts close to cage sites: most of the criteria used to evaluate site applications address the likelihood of local benthic impacts at a site; operational monitoring has concentrated on measures of change in the local benthic community through tools like Eh and sulphide measurements, video surveys, and the characterization of changes in the local benthic community structure. There is some implicit consideration of cumulative, wider-scale impacts in site applications through tools like the decision support system (Hargrave, 2002) which, although it does not specifically consider the potential for eutrophication, does include a consideration of the closeness of other farms, other industry, marine protected areas, endangered populations etc within a few kilometers of the proposed site. A similar focus on local benthic impacts has been common in other areas. Levings et al. (1995) reviewed siting criteria in eight different jurisdictions. At that time, large scale eutrophication was only explicitly considered in Norway, which included water quality on a fjord-wide basis as a siting criterion; in other places, the emphasis was on the potential for the formation of a debris pile under the cages or for impacts on other resources within a few hundred metres. More recently, Scotland has adopted regulations that explicitly consider the potential for eutrophication on large scales (whole sea lochs, Gillibrand et al., 2002). In Tasmania, it has recently been recognized that models of nutrient dynamics and eutrophication must be part of management planning in areas of high density farming activity (Crawford, 2003). As shown in Fig. 6, only a very small fraction of the total waste stream from salmon farms is found in the debris piles under cages, even in an area like Lime Kiln Bay in SWNB, which, based on benthic impacts, was considered to be a highly depositional environment. Furthermore, the above calculations show that given the necessary combination of waste inputs and sensitivity of the receiving environment, finfish cage aquaculture can impact areas much larger than the farms themselves, on scales of kilometers to tens of kilometers. Siting criteria must consider the potential for cumulative, far-field eutrophication impacts from marine finfish farms. When any threat for widespread impacts from eutrophication exists, operational requirements for monitoring must also reflect this reality. A failure to consider the potential for large scale eutrophication may be ignoring one of the most significant potential impacts of marine finfish culture.

As described above, evaluating the potential for eutrophication caused by marine finfish aquaculture requires assessments of both the waste inputs and the carrying capacity of the receiving environment. A combination of data on farm operations and mass balance models is the only currently available tool for assessing waste inputs. Information on fish numbers and/or biomass and feed use are required for all such calculations. To put this point into language current in discussions of marine environmental quality, fish numbers are probably the most, and may be the only, useful indicator of far-field eutrophication impacts on ecosystem quality. The need for information on fish numbers and feed use for assessing eutrophication impacts may be at odds to some approaches to performance-based standards, in which industry would trade control over fish numbers and other details of farm operation for guaranteeing minimum environmental standards in the receiving environment.

The second aspect of evaluating the potential for eutrophication, assessing the carrying capacity of the receiving environment, requires a context. Value judgements on whether ecosystem changes are beneficial or harmful and how much change is acceptable must be made before appropriate measures of carrying capacity can be selected and limits set. Explicit objectives for integrated coastal management are useful here. Whether eutrophication is considered harmful or not depends on the goal set: minimum disturbance of the receiving environment, maximum disturbance compatible with a sustainable industry, or somewhere in between. Once the goal is set, the selection of a means of assessing aquaculture impacts becomes more obvious: e.g. if the objective is to leave the ecosystem in a nearly pristine state, then perhaps aquaculture should only be permitted to modify natural nutrient cycling by a small amount; if the objective is to avoid an ecosystem that is too ‘eutrophic’, but is naturally not very productive to start with, then perhaps some considerable amount of eutrophication is acceptable. Managing for larger scale impacts poses some additional challenges, such as the need to allocate portions of the available carrying capacity to different users. Exactly how carrying capacity should be assessed in any given case, and what acceptable limits are, are points that are very much in development. Comparing aquaculture discharges to other inputs or comparing nutrient cycling by aquaculture with that of the natural ecosystem as has been done here are possible starting points. In the Scottish scheme (Gillibrand et al., 2002), a classification scheme for different sea lochs is developed based on combined benthic and eutrophication impacts, but the boundaries between categories are fuzzy: lochs are more or less favourable, but aquaculture development is not expressly forbidden at any level.

One of the attractions of basing aquaculture management on local benthic impacts is the relative ease with which such impacts can be predicted, measured and the severity of those impacts judged: e.g. few would argue that sediments made anoxic by farming wastes were environmentally acceptable. Furthermore, there is little doubt that a local farm is the cause of a particular debris field. However, as the scale of the impact increases, it becomes more difficult to both predict and measure the impacts, and to associate impacts with a given farm site. In the absence of clear-cut cause-effect relationships, management must recognize that scientific certainty will not be available, and adopt a suitably precautionary approach. Silvert and Cromey (2001) distinguished between the primary effects of aquaculture, such as nutrient enrichment which are relatively easy to predict or to measure, and secondary effects, which are the resulting changes to the ecosystem. The secondary effects are the ones that really matter. They made the statement " … *modelling secondary effects … may prove impossible … The only realistic alternative may be to set arbitrary levels by negotiations with the various 'stakeholders', with little solid scientific input to the process.*" While the situation may not be this bleak, such an approach involving all interested parties in the process of making decisions based on incomplete information forms a basis for dealing with the current scientific uncertainty and has been a successful strategy in some areas of resource management.

Finally, how does the state of knowledge compare with the information necessary to manage marine finfish aquaculture with respect to potential eutrophication impacts? I believe that some minor improvements can be made by increasing our knowledge of the fate of the aquaculture waste (such as better quantitation of the amount of feed wasted, the wastes in debris piles and the role of the fouling communities) and by tuning models to better reflect current industry practices, but that there are significant improvements that can be made to our understanding and ability to predict secondary impacts, whether they be changes in overall productivity or production patterns, the occurrence of harmful algal blooms, or other changes to the structure and functioning of the ecosystem, and to propose meaningful limits for eutrophication impacts. These advances will require further research, and a management approach flexible enough to adapt to both this new scientific understanding as it becomes available and to the inevitable changes in the way the industry operates.

REFERENCES

Ackefors, H, and M. Enell. 1994. The release of nutrients and organic matter from aquaculture systems in Nordic countries. J. Appl. Ichthyol. 10: 225-241.

Bricker, S.B., Clement, C.G., Pirhalla, D.E., Orlando, S.P. and D.R.G. Farrow. 1999. National estuarine eutrophication assessment: Effects of nutrient enrichment in the nation's estuaries. Special Project Office and the National Centers for Coastal Ocean Science, Nation Oceanic and Atmospheric Administration, Silver Spring, MD. 71 p.

Bugden, J.B.C., Hargrave, B.T., Strain, P.M. and A.R.J. Stewart. 2001. Spatial patterns of some physical and chemical variables in Passamaquoddy Bay and Letang Inlet, southwestern Bay of Fundy, September 1999. Can. Tech. Rep. Fish. Aquat. Sci. 2356: iv + 96 p.

Chamberlain, J., D. Stucchi, L. Lu and C. Levings. 2005. The suitability of DEPOMOD for use in the management of finfish aquaculture sites, with specific reference to Pacific Region. DFO Can. Sci. Advis. Sec. Res. Doc. 2005/035.

Cho, C.Y., Hynes, J.D., Wood, K.R. and H.K. Yoshida. Development of high-nutrient-dense, low-pollution diets and prediction of aquaculture wastes using biological approaches. Aquaculture 124: 293-305.

Crawford, C. 2003. Environmental management of marine aquaculture in Tasmania, Australia. Aquaculture 226: 129-138.

Danish Environmental Protection Agency. http://www2.dmu.dk/1_Viden/2_Miljoe-tilstand/3_vand/4_eutrophication/definition.htm#eutrophication

Hargrave, B.T. 2003. Far-field environmental effects of marine finfish aquaculture. In Fisheries and Oceans Canada: A scientific review of the potential environmental effects of aquaculture in aquatic ecosystems. Volume I. Can. Tech. Rep. Fish. Aquat. Sci. 2450: ix + 131 p.

Harrison, W.G, Perry, T. and W.K.W. Li. 2005 (*In press*). Ecosystem indicators of water quality. I. Plankton biomass, primary production and nutrient demand. Chapter 3 in: The Handbook of Environmental Chemistry. Volume 5: Water Pollution. Environmental Effects Of Marine Finfish Aquaculture. Springer, Berlin Heidelberg New York.

EEA, European Environmental Agency. 1999. Nutrients in European ecosystems. Environmental assessment report No 4.

EEB, European Environmental Bureau. 2001. EEB Handbook on EU Water Policy under the Water Framework Directive.

Gillibrand, P.A., Gubbins, M.J., Greathead, C., Davies, I.M. 2002. Scottish executive locational guidelines for fish farming: predicted levels of nutrient enhancement and benthic impact. Scottish Fisheries Research Report Number 63 / 2002 Fisheries Research Services, Marine Laboratory, Aberdeen.

Hargrave, B.T. 2002. A traffic light decision system for marine finfish aquaculture siting Ocean & Coastal Management, 45: 215-235 .

Levings, C.D., Ervik, A., Johannessen, P,. and J. Aure. 1995. Ecological criteria used to help site fish farms in fjords. Estuaries 18: 81-90.

Newton, A., Icely, J.D., Falcao, M, Nobre, A., Nunes, J.P., Ferreira, J.G. and C. Vale. 2003. Evaluation of eutrophication in the Ria Formosa lagoon, Portugal. Cont. Shelf Res. 23: 1945-1961.

Nixon, S. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. Ophelia 41: 199-219.

OSPAR. 1998. OSPAR Strategy to Combat Eutrophication. Reference number: 1998–18.

Page, F.H., Losier, R., McCurdy, P., Greenberg, D., Chaffey, J. and B. Chang. 2005 (In press). Dissolved oxygen and salmon cage culture in the southwestern New Brunswick portion of the Bay of Fundy. Chapter 1 in: The Handbook of Environmental Chemistry. Volume 5: Water Pollution. Environmental Effects Of Marine Finfish Aquaculture. Springer, Berlin Heidelberg New York.

Robinson, S.M.C., Auffrey, L.M. and M.A. Barbeau. 2005 (*In press*). Far-field impacts of eutrophication on the intertidal zone in the Bay of Fundy with emphasis on the soft-shell clam, *Mya arenaria*. Chapter 13 in: The Handbook of Environmental Chemistry. Volume 5: Water Pollution. Environmental Effects Of Marine Finfish Aquaculture. Springer, Berlin Heidelberg New York.

Schaanning, M.T. and P. Kupka Hansen. 2005 (*In press*). The suitability of simple electrode measurements for assessment of benthic organic impact and their use in a management system for marine fish farms. Chapter 18 in: The Handbook of Environmental Chemistry. Volume 5: Water Pollution. Environmental Effects Of Marine Finfish Aquaculture. Springer, Berlin Heidelberg New York.

Schramm. W. and Nienhuis, P.H. (1996) Marine Benthic Vegetation: Recent changes and the effects of eutrophication. Springer-Verlag, Berlin. xix + 470 pp.

Silvert, W. and C.J. Cromey. 2001. Modelling impacts. p. 154-181 *In*: Environmental impacts of aquaculture. K.D. Black [ed]. Sheffield Academic Press, Sheffield, UK.

Strain, P.M. 2003. Appendix A. Oxygen demand and nutrient loads from finfish aquaculture in southwestern New Brunswick. *In*: Dept. of Fisheries and Oceans.

Salmon holding capacity in southwestern New Brunswick. Can. Tech. Rep. Fish. Aquat. Sci. 2489: iv + 60 p.

Strain, P.M. and B.T. Hargrave. 2005 (*In press*). Salmon aquaculture, nutrient fluxes and ecosystem processes in southwestern New Brunswick. Chapter 2 in: The Handbook of Environmental Chemistry. Environmental Effects Of Marine Finfish Aquaculture. Volume 5: Water Pollution. Springer , Berlin Heidelberg New York.

Strain, P.M., Wildish, D.J. and Yeats, P.A. 1995 The application of simple models of nutrient loading and oxygen demand to the management of a marine tidal inlet. Mar. Poll. Bull. 30: 253 - 261.

Table 1. Waste outputs per tonne of fish produced (expressed as kg of C, N, P, and O_2). Data for SWNB from Strain and Hargrave (2005). Results are for Atlantic salmon unless otherwise noted.

kg / tonne of fish	Carbon	Nitrogen	Phosphorus	Oxygen
SWNB				
Dissolved wastes	226	33	4.9	-590
Buried in sediments	76	9	2.3	-
Total	**302**	**42**	**7.2**	**-590**
Literature				
Scotland: Gillibrand et al., 2002				
Dissolved wastes[1]		42		
Buried in sediments[1]		6.3		
Total		**48.2**		
Norway: Bergheim et al., (1991)		**90**	**10**	**-500**[2]
Nordic Countries: Ackefors and Enell (1994)		**60**	**10**	
Laboratory: Cho et al., 1994 (Brown trout and lake trout)		**46 - 65**	**6.2 – 10.5**	
SWNB: Strain et al. (1995)		**66**	**10.5**	**-345**[2]

[1] Dissolved and solid waste estimates from Gillibrand et al. (2002) adjusted for rapid decomposition as described in the text for SWNB.

[2] This estimate does not include the O_2 consumed by fish respiration

Table 2. Deposition and sedimentation rates predicted from mass balance model for calendar year 2 of the salmon grow-out cycle in SWNB. Calculations are based on a farm site with 80,000 fish in 9 polar circle pens (circumference = 60 m, depth = 6 m) yielding a density 16.9 kg m^{-3} of pen volume or 101 kg m^{-2} of pen area. Deposition rates are given for the day of maximum discharge (Oct 2), at which time the fish weighed 3.27 kg. Sedimentation rates are averaged over the entire calendar year.

	Spreading factor (area of pens : area of impact)	**C**	**Mass (dry matter)**	**N**	**P**	**Sedimentation rate (cm y^{-1})**
Maximum deposition (kg $farm^{-1}$ d^{-1})		245	490	29	7.4	
Maximum sedimentation (g m^{-2} d^{-1})	1:2	48	95	5.6	1.44	2.9
Maximum sedimentation (g m^{-2} d^{-1})	1:10	9.5	19.0	1.12	0.29	0.58
Maximum sedimentation (g m^{-2} d^{-1})	1:50	1.90	3.80	0.22	0.057	0.116
Maximum sedimentation (g m^{-2} d^{-1})	1:100	0.95	1.90	0.11	0.029	0.058

Table 3. Concentration changes predicted for different coastal management regions in SWNB (see Fig. 8) in µmoles / liter. Data from Strain and Hargrave (2005). These concentration changes are those predicted for the maximum daily discharges (in October) during the year of the odd/even year cycle during which fish biomass is highest.

Substance	Northern Passamaquoddy Bay	Letang Inlet	Deer and Campobello Islands
Nitrogen	0.18	5.5	0.18
Phosphorus	0.012	0.38	0.012
Dissolved O_2	-1.4	-43	-1.4
Residence Times (days)	15	9	3

Table 4. Comparison between oxygen, nitrogen and carbon processing by salmon farms in SWNB with processing of those elements by natural processes. Aquaculture estimates are based on site allocations in 2002 and the year of the year class cycle with higher biomasses in each CMR. Numbers in the table are the ratio of element cycling by salmon aquaculture to that by natural processes, expressed as a percentage: e.g. in Lime Kiln Bay, 3.3 times as much oxygen is processed by salmon aquaculture as by natural processes. Data from Strain and Hargrave (2005).

CMR	Oxygen	Nitrogen	Carbon
Passamaquoddy Bay	0.08	3.2	2.1
Letang Inlet	2.6	70	34
Lime Kiln Bay	20	330	160
Bliss Harbour	9.1	210	100
Back Bay	13	170	84
Letete Passage	0.34	11	5.2
Campobello / Deer Islands	0.40	16	10
Grand Manan Island	0.28	11	7.1
Cobscook Bay	0.77	8.1	4.5

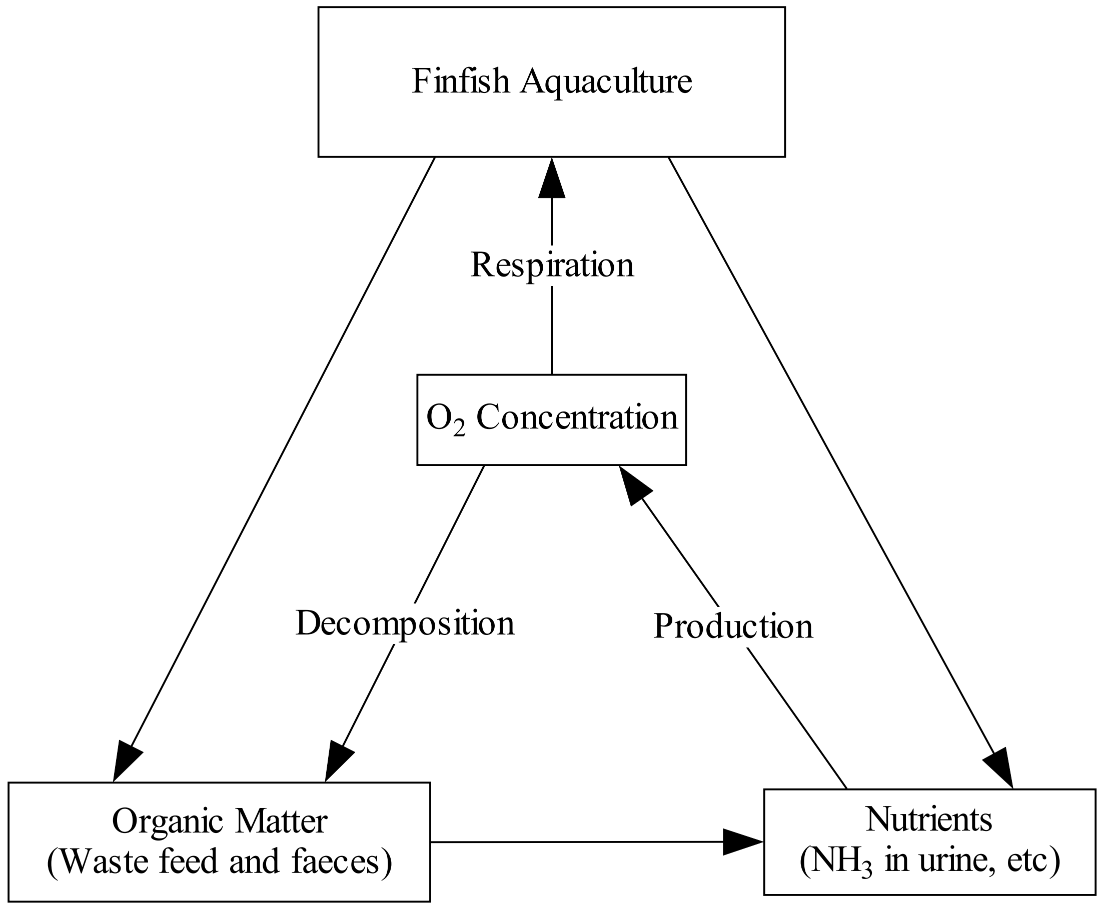

Fig. 1. The interaction of finfish wastes with the cycling of energy and nutrients in the ecosystem. Arrow into boxes indicate increases in that ecosystem reservoir. Aquaculture adds organic matter and inorganic nutrients to the natural metabolism of the ecosystem, both of which contribute to eutrophication. Finfish respiration also puts a demand on the available dissolved oxygen in the ecosystem, which directly impacts this symptom of eutrophication.

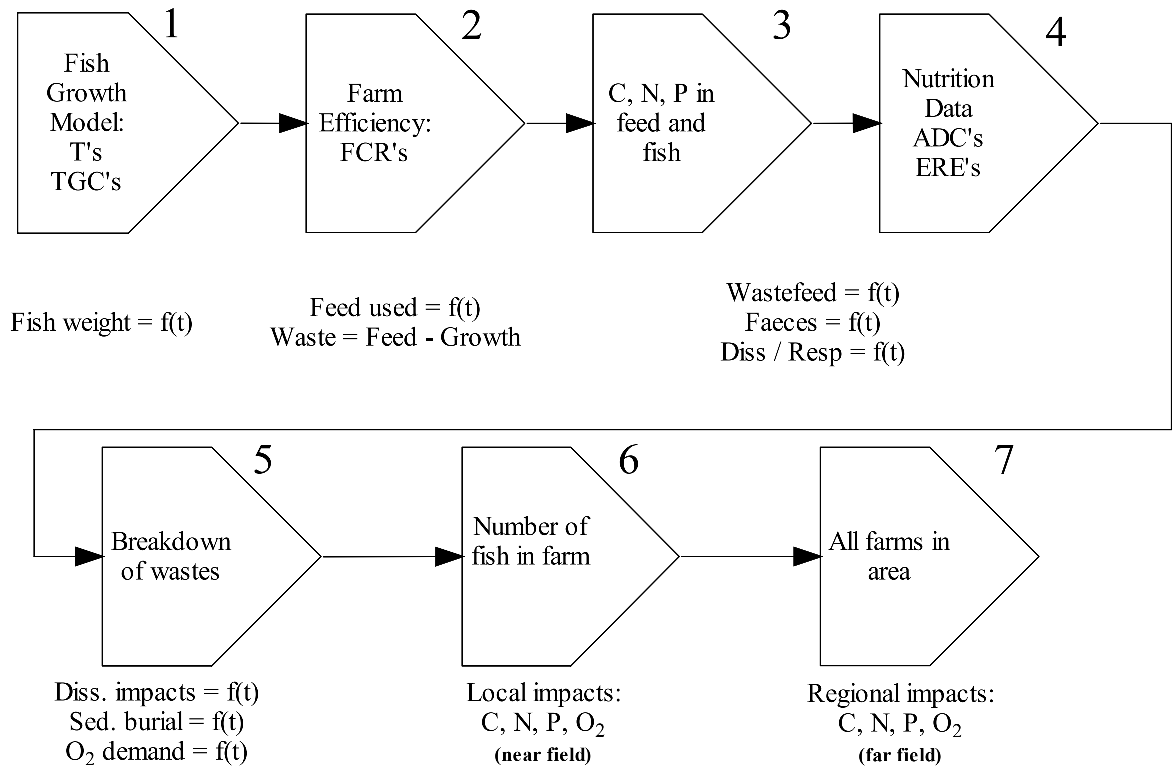

Fig. 2. The sequence of calculations used to estimate eutrophication wastes from net pen aquaculture. The information added at each step is shown inside the symbols; the output of that step is shown underneath the symbol. For example, in step 1, a fish growth model is combined with the seasonal temperature cycle and thermal growth coefficients (TGC's) to predict the size of individual fish as a function of time in the grow-out cycle. FCR = feed conversion ratio; ADC = apparent digestibility coefficient; CRE = carbon retention efficiency; C,N,P = Carbon, Nitrogen, Phosphorus; Diss. = dissolved; Resp. = respiration; Sed. = sediment.

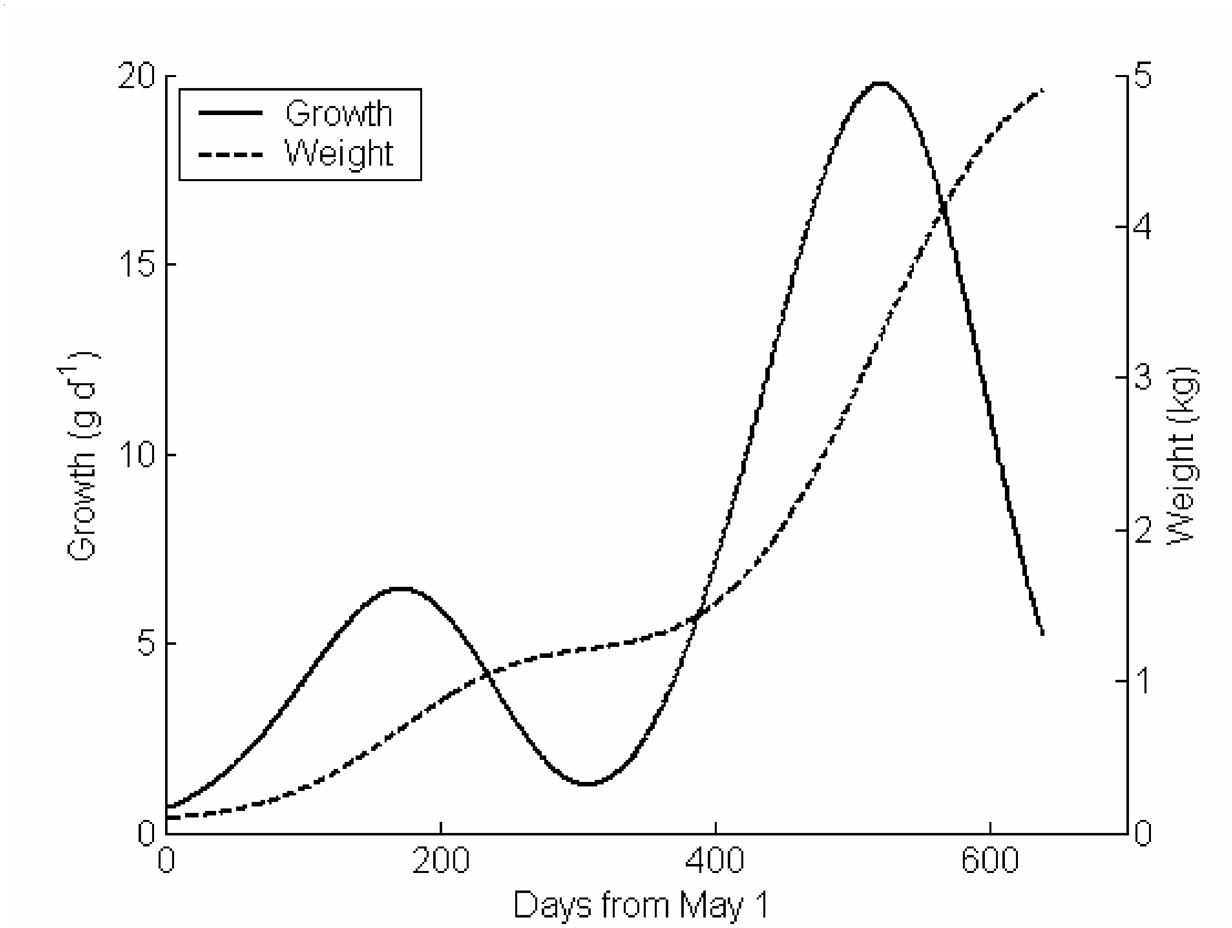

Fig. 3. The size (kg) and growth rate (g d^{-1}) of a single salmon over its grow-out cycle in SWNB conditions predicted by a thermal growth coefficient model. (Strain and Hargrave, 2005).

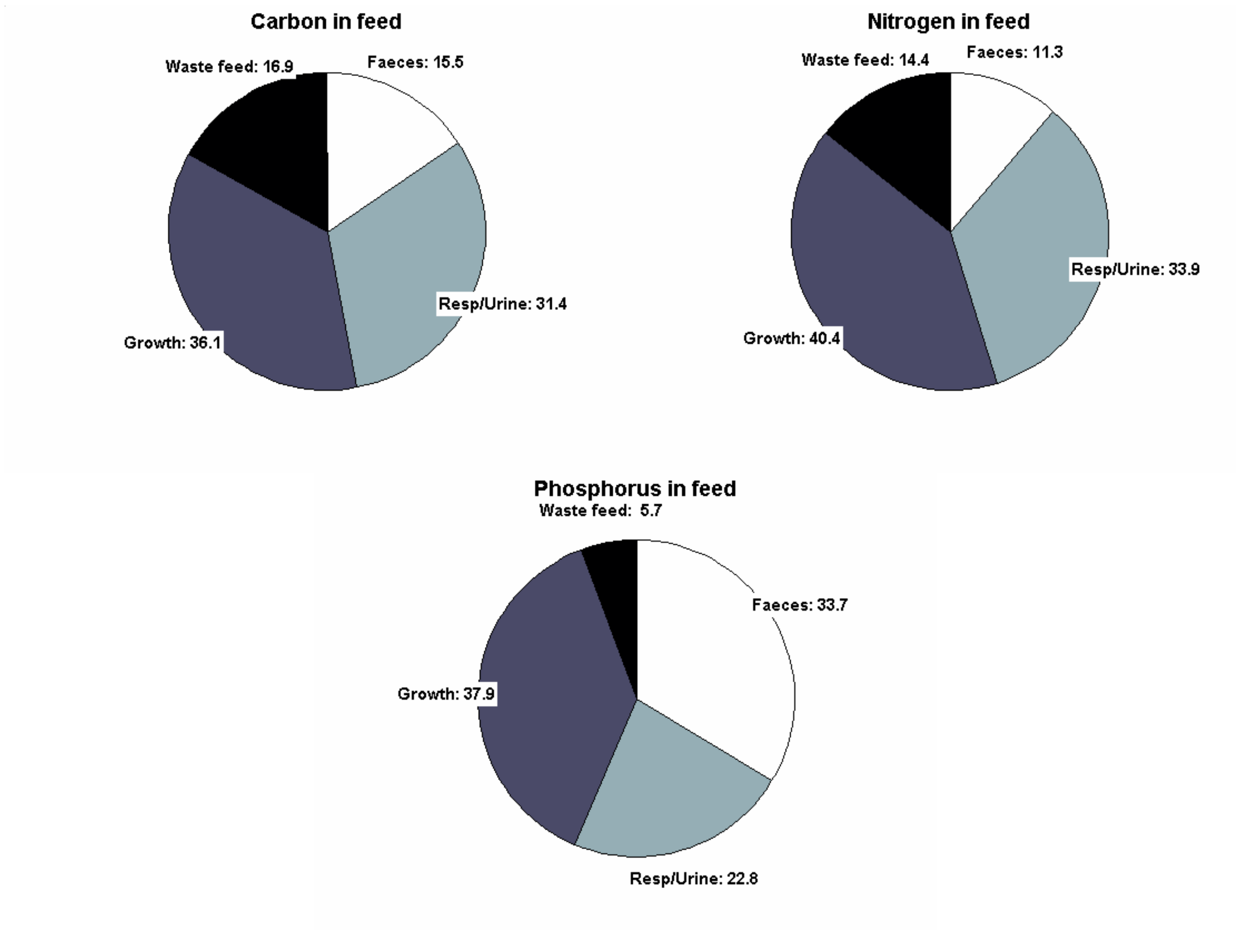

Fig. 4. The initial fractionation of feed used on salmon farms in SWNB. These numbers show how feed is initially separated into growth (sometimes termed retained carbon, nitrogen or phosphorus), metabolic consumption (respiration for carbon, urine for nitrogen and phosphorus), particulate faeces and waste feed as fish are fed on farm sites.

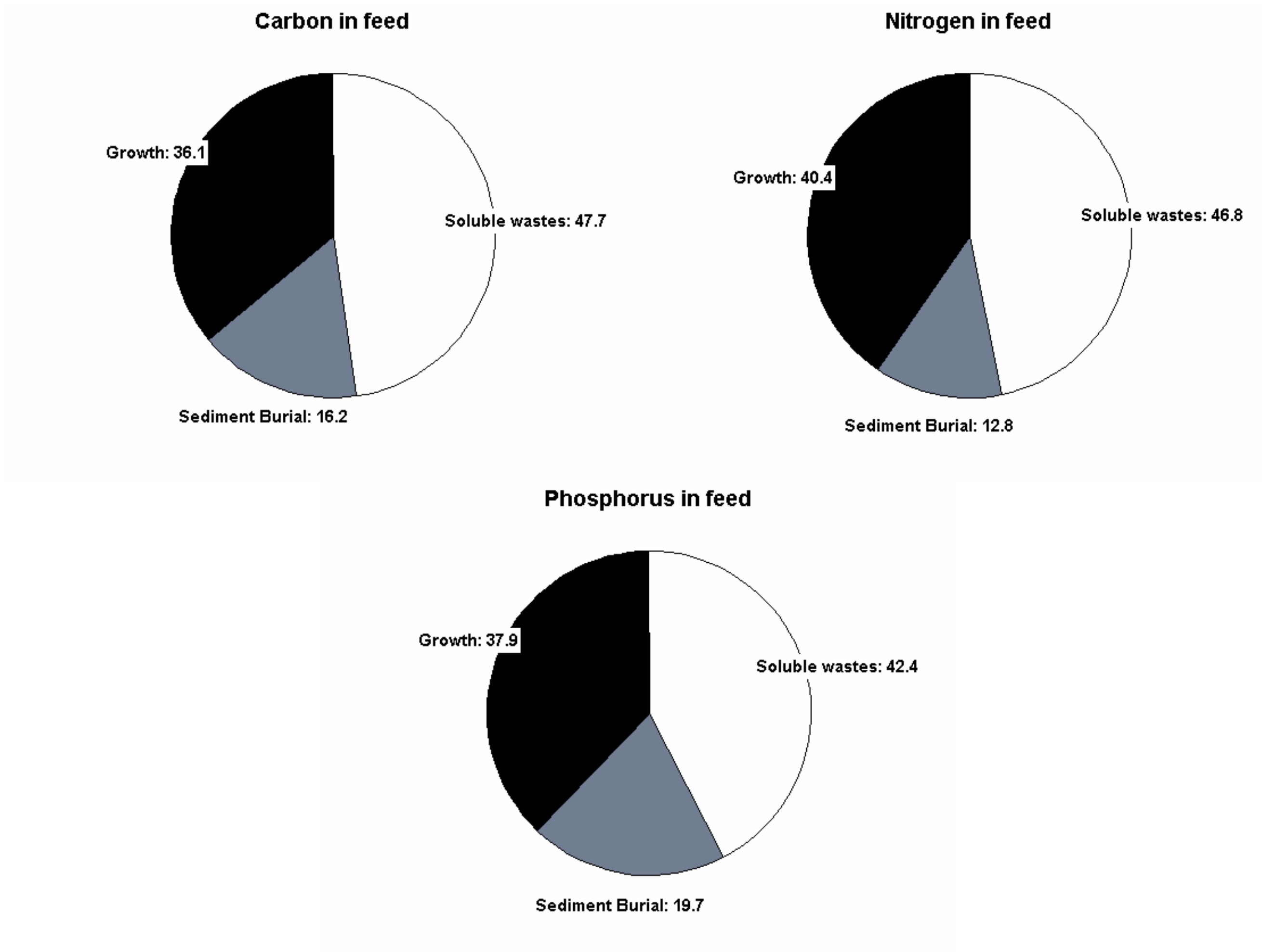

Fig 5. The fate of feed used on salmon farms in SWNB. These numbers are adjusted from the initial breakdown shown in Fig. 4 to account for that fraction of the particulate wastes that are labile and will be decomposed quickly enough to impact conditions in the local area. The net effect of this decomposition is the conversion of solid phase organic nitrogen and phosphorus to dissolved inorganic nitrogen and phosphorus.

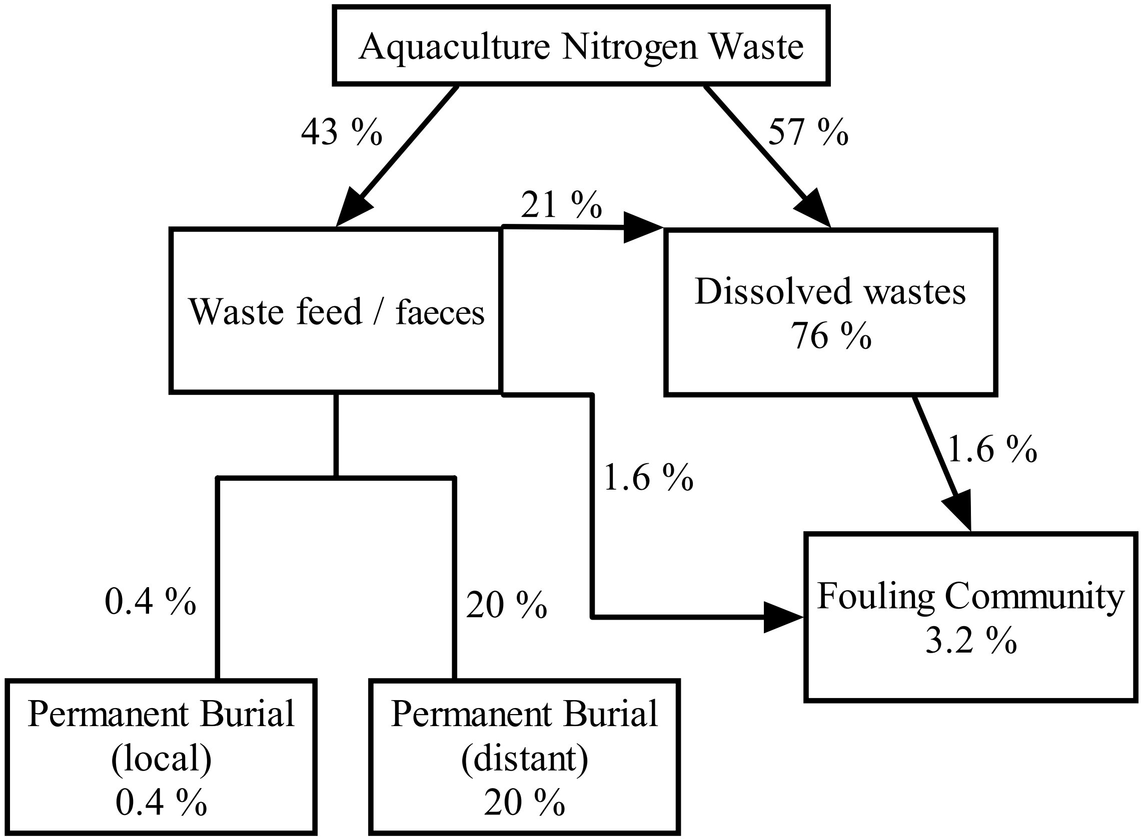

Fig. 6. The fate of nitrogen wastes discharged from salmon farms in SWNB. Numbers on connecting lines represent percentage of the total nitrogen discharge that is transferred from one reservoir to another. Numbers in boxes represent the percentage of the total discharge in each reservoir. The nitrogen supply for the fouling community has been arbitrarily divided equally between particulate and dissolved sources.

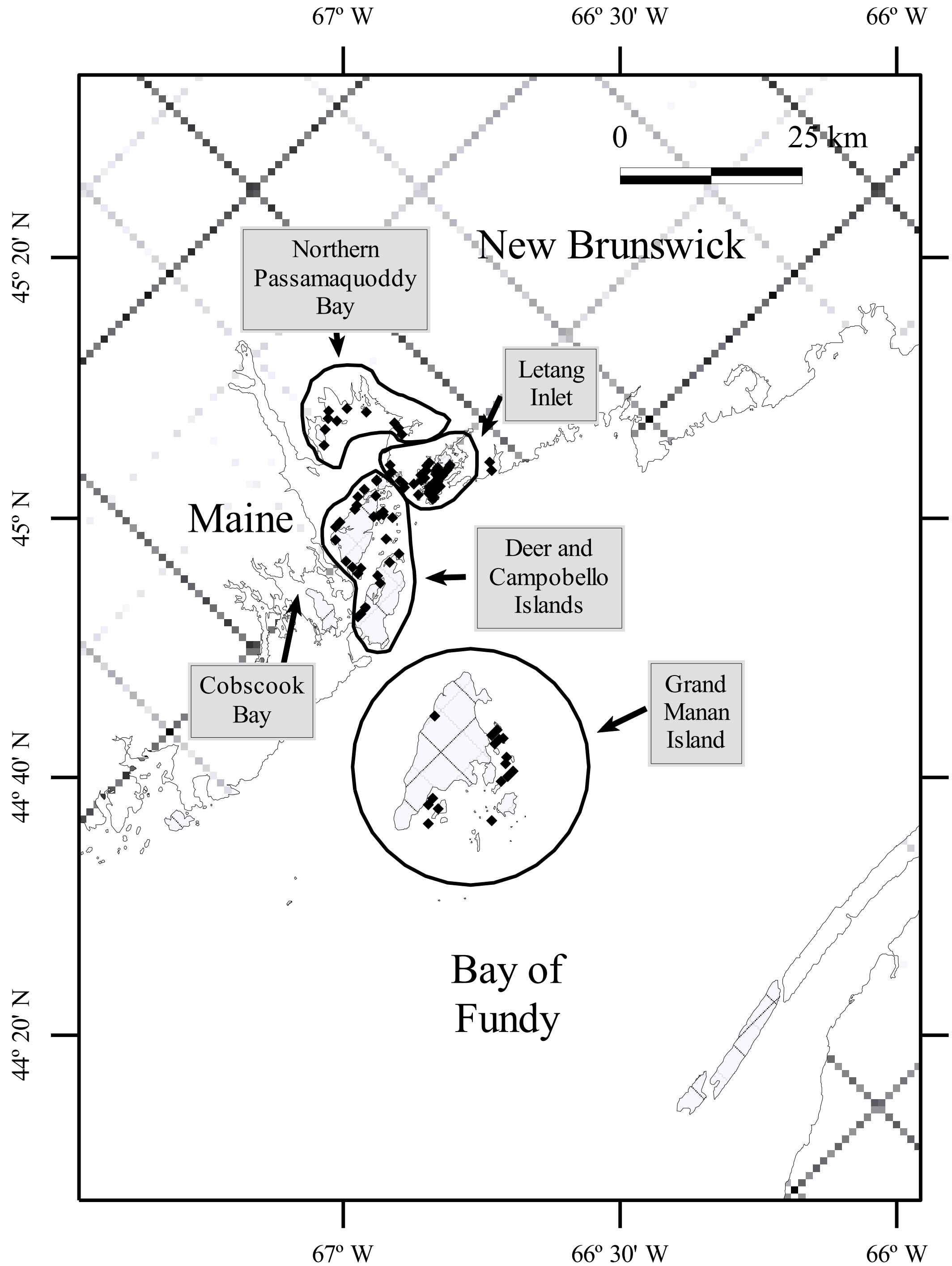

Fig. 7. The salmon farming region in the Bay of Fundy, southwestern New Brunswick. Farm sites licensed in 2002 are indicated by diamonds. Adapted from Strain and Hargrave (2005). The Coastal Management Regions (CMR's) discussed in the text are the regions denoted by solid lines.

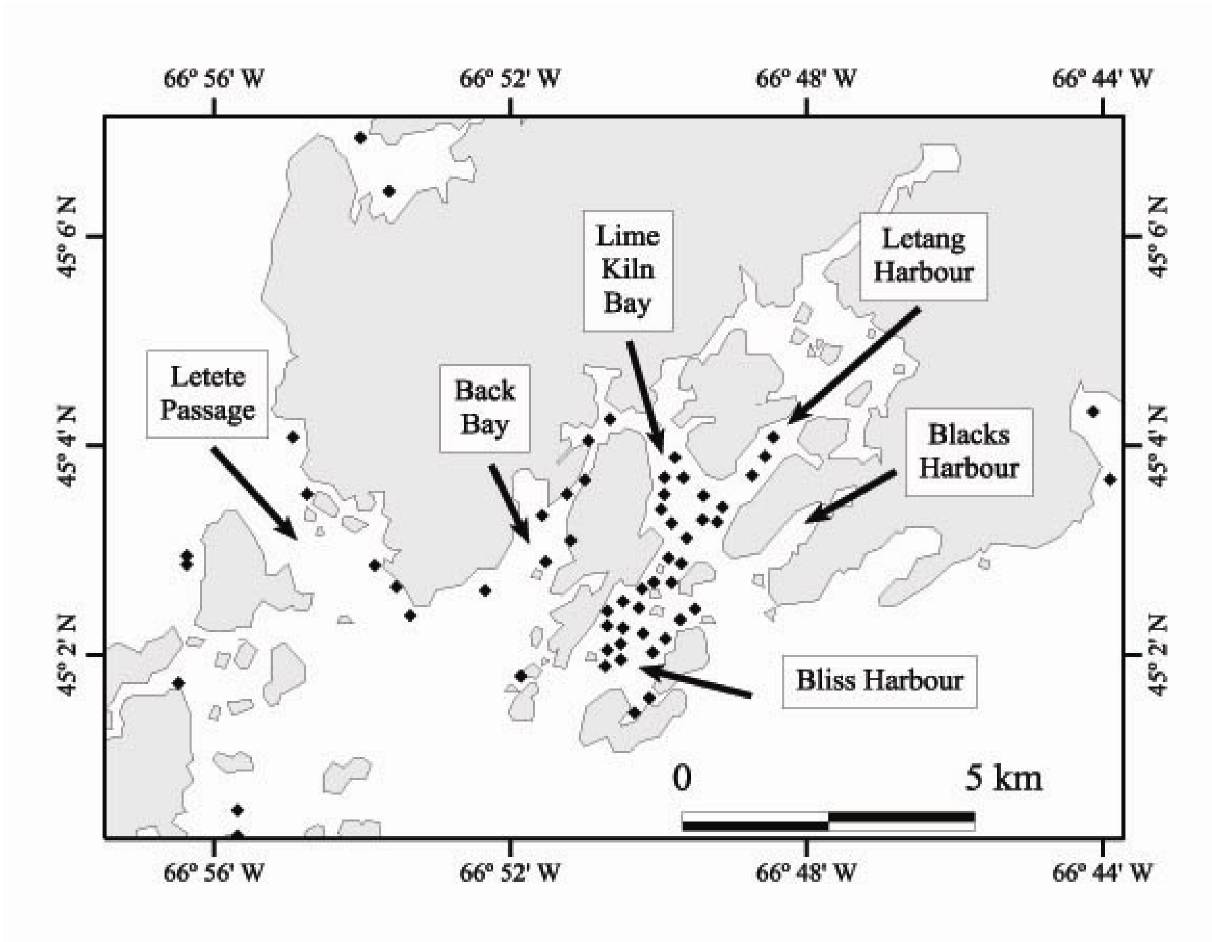

Fig. 8. Salmon farms in the Letang Inlet, SWNB, in 2002. Farm sites are indicated by diamonds. Adapted from Strain and Hargrave (2005).

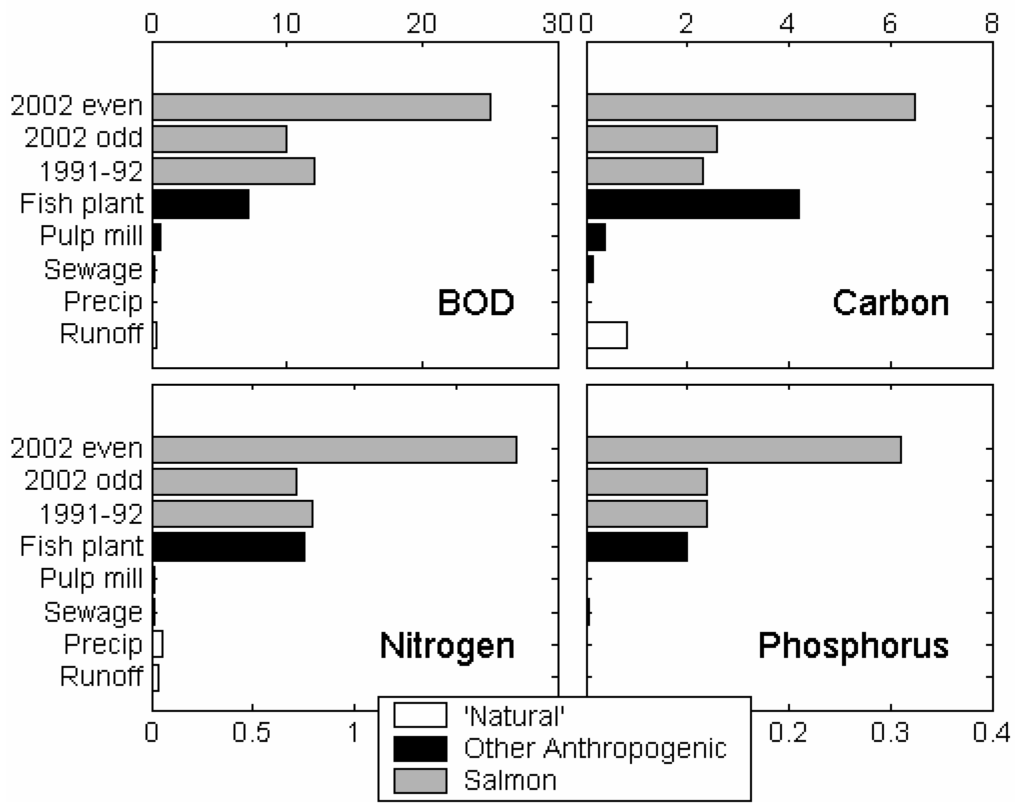

Fig. 9. Discharges into Letang Inlet, SWNB (metric tonnes per day). Estimates for the salmon industry in 2002 (data from Strain and Hargrave, 2005) are based on the licensed capacity of the Letang Inlet. Discharges vary between odd and even years in 2002 due to single year class practices in the Letang at this time. Other estimates are based on original data from Strain et al. (1995), updated by Strain and Hargrave (2005).

Appendix. Mass Balance Calculations and Associated Uncertainties

The case study calculations described above for the salmon industry in Southwestern New Brunswick (SWNB) are based on a salmon growth model and a mass balance approach that combines the salmon growth predictions with data on feed use, feed nutrition and geochemistry to predict dissolved and particulate waste discharges of carbon, nitrogen and phosphorus, and oxygen demand from salmon net-pen operations. For a more detailed description of these calculations, explanations for the choices made for parameter values and appropriate references, see Strain and Hargrave (2005). These calculations have been implemented in a Microsoft Excel file which is available from the author. This Appendix lists the steps in the calculation and does an analysis of the uncertainties in the results based on the uncertainties in the inputs.

Mass Balance Calculations

The steps in the calculation are listed below. Step numbers correspond to the steps in Fig. 2.

1. The daily weight and growth of the salmon are calculated from a model based on a thermal-unit growth coefficient (TGC):

$$W_f = \left[W_i^{1/3} + \sum\left(TGC*T*t\right)\right]^3$$

   where:
   W_i, W_f = initial, final body weights (g wet weight)
   TGC = thermal-unit growth coefficient
   T = temperature
   t = time (days)

   Growth is given by:

   growth = W_f - W_i

   The model was run using 1 day time steps, daily water temperatures, and a TGC value 0.00265 $g^{1/3} \cdot {}^{o}C^{-1} \cdot d^{-1}$.

2. Next the amount of feed used is estimated. For SWNB, an industry wide average of the feed conversion ratio (FCR = feed used (wet weight) / fish produced (whole weight)) was used to estimate daily feed used for each fish:

   feed = growth * FCR

   The waste is then simply:

   waste = feed – growth

3. In this step, the amount of carbon, nitrogen, and phosphorus in feed and fish are estimated so that the waste may be calculated separately for C, N and P:

   The C and N contents of the feed may be estimated from the weight of the feed using standard biochemical formulae that calculate C and N from the proximate composition provided by feed manufacturers (converted to a dry weight basis, if necessary):

   $$C = 0.5 * f_{prot} + 0.7 * f_{fat} + 0.4 * f_{carbohydrate}$$

   $$N = 0.16 * f_{prot}$$

   where
   C, N = fraction (by mass) of carbon, nitrogen in dry feed
   f_{prot}, f_{fat}, $f_{carbohydrate}$ = fractions of protein, fat (lipid) and carbohydrate (fibre) in dry feed

   The percentage phosphorus in feed was directly available from the feed manufacturers.

   The C content in fish was estimated from the proximate composition and the average water content of salmon reared in SWNB. Direct measurements of N and P in whole salmon are available. The total C, N or P in waste can then be estimated from equations like:

   $$C_{waste} = C_{feed} - C_{retained}$$

   where $C_{retained}$ is the carbon retained in fish tissue.

4. In this step, estimates are made of the C, N and P in each waste fraction:

   The waste C discharge includes the carbon consumed by fish respiration as well as the organic carbon in fish faeces and uneaten feed. The N and P discharges include the N and P excreted by the salmon as dissolved wastes (urine) and faeces, and the nitrogen and phosphorus in the uneaten feed. The carbon fractions of the waste may be written:

   $$C_{waste} = C_{respiration} + C_{wastefeed} + C_{faeces}$$

   $C_{respiration}$ and C_{faeces} may be estimated from apparent digestibility coefficients (ADC) and carbon (or energy) retention efficiencies (CRE) available from the fish nutrition literature for carbon (or energy). These are defined as:

   $$ADC = 1 - \frac{C_{faeces}}{C_{ingested}} \qquad CRE = \frac{C_{retained}}{C_{respiration} + C_{retained}}$$

   Rearranging the *ADC* and *ERE* equations leads to the following expressions:

$$\frac{C_{respiration}}{C_{retained}} = \frac{1 - ERE}{ERE} \qquad \frac{C_{faeces}}{C_{retained}} = \frac{1 - ADC}{ADC * ERE}$$

Now that $C_{respiration}$ and C_{faeces} are known, and C_{waste} was determined in step 3, $C_{wastefeed}$ can be calculated by difference:

$$C_{wastefeed} = C_{waste} - C_{respiration} - C_{faeces}$$

Similar expressions are used for N and P, with urine output of N and P substituting for the respired C.

The oxygen demand of respiration is calculated from the respired carbon and the stoichiometry of respiration:

$$O_{2\ respiration} = C_{respiration} * 32 / 12$$

At this point in the calculations, estimates for the carbon, nitrogen and phosphorus in metabolic wastes (respiration or urine), faeces and waste feed and the O_2 demand of respiration are available for a fish for each day of the grow-out period.

5. A fraction of the waste feed and faeces are very labile and will decompose quickly, converting particulate forms of C, N and P to dissolved forms, and consuming oxygen. The total dissolved wastes are increased by the decomposition of the solid wastes; the solid wastes available for burial in the sediments are reduced. From estimates of the labile fraction in the solid wastes, the dissolved wastes and particulate wastes available for burial in the sediments can be estimated as shown in these equations for N:

$$N_{dissolved} = N_{urine} + f_{wastefeed} * N_{wastefeed} + f_{faeces} * N_{faeces}$$

$$N_{part} = (1 - f_{wastefeed}) * N_{wastefeed} + (1 - f_{faeces}) * N_{faeces}$$

where $f_{wastefeed}$, f_{faeces} are the labile fractions of waste feed and faeces.

Similar equations are used for C and P.

The total O_2 demand is increased by this process, and is estimated from:

$$O_{2\ demand} = O_{2\ respiration} + f_{wastefeed} * C_{wastefeed} * 32 / 12 + f_{faeces} * C_{faeces} * 32 / 12$$

6. After step 5, we have estimates of feed used, metabolic wastes, waste feed, faeces and the dissolved and particulate waste streams for a single fish for each day of the grow-out cycle as C, N and P, and of the O_2 consumed by respiration and waste decay. These wastes can be summed over the entire grow-out cycle to determine the wastes per ton of fish produced (see Table 1), or the seasonal maxima in these wastes can be

determined and expressed per 1000 fish at a farm, as is done in Table 2 of Strain and Hargrave (2005).

7. Bay wide estimates of impacts can also be determined from these results, by adding up the impacts of all the fish in a bay, making allowances for the numbers of fish in each year class present in the bay.

Uncertainties in Mass Balance Calculations

As developed and applied in SWNB, some 23 parameters go into these models for the predictions of waste discharges from individual farms. For models with large numbers of parameters, Monte Carlo statistics provide a convenient way to estimate uncertainties in model outputs due to uncertainties in the inputs. The Monte Carlo analysis requires estimates of the distribution of uncertainties in the inputs. Typically, normally distributed errors in the inputs are assumed and their variances estimated. The model is then run repeatedly using input parameters chosen randomly from their distributions. Statistics on the model outputs can then be used as estimates of reliability. Some particular features associated with the finfish waste models described here required special treatment:

- Interdependencies between parameters. For example, the sum of $f_{protein}$, f_{fat}, $f_{carbohydrate}$ and f_{ash} in feed must equal one. In the Monte Carlo analysis, the surplus (deficit) was distributed over all four fractions after these four parameters were randomly selected for each run of the model.
- The model sometimes 'fails'. Some combinations of parameters do not contain enough carbon, nitrogen or phosphorus to grow the fish. These model runs were not included in the error analysis statistics.
- Using normally distributed errors in the inputs can lead to inappropriate values for the parameters (eg fractions less than 0 or greater than 1). Minimum or maximum values were specified to prevent such occurrences. Any time a randomly selected parameter exceeded these extremes, it was set equal to its minimum or maximum value for that model run.

Table A.1 lists the values of the input parameters used in the SWNB calculations and estimates of their standard deviations used in the Monte Carlo error analysis. Tables A.2 and A.3 list the values of some of the model outputs (based on the input parameters in Table A.1) and their coefficients of variation as determined from 2000 randomly selected sets of input parameters, 1363 of which were successful and produced model output. Table A.2 lists estimates of the reliability of outputs expressed per ton of fish produced; Table A.3 lists estimates of the reliability of outputs expressed as maximum discharges during calendar year 2 of the grow-out cycle, per 1000 fish. This analysis has been performed on the dissolved and particulate ('part') waste streams that includes the decay of waste feed and faeces, as well as the total of the dissolved and particulate wastes that are independent of the values of $f_{wastefeed}$ and f_{faeces}.

The Monte Carlo analysis estimates the overall precision of the model outputs, but it does not indicate which input parameters contribute most to the uncertainty of the outputs. For each model output, the relative influence of the input parameters was determined by calculating the percentage change in the output that would correspond to a 1 σ change in that input, while the remaining inputs were maintained at the values in Table A.1. Tables A.2 and A.3 also list the three most influential inputs for each output.

For model outputs expressed per tonne of fish produced, the coefficients of variation are between 12 and 52 %. Uncertainties in the predicted dissolved wastes (i.e. the water column impacts) range from 13 – 15 %; uncertainties in predicted particulate wastes (i.e. the sediment impacts) range from 24 – 52 %. Clearly, the dissolved wastes are better predicted by the model than the particulate wastes. Because dissolved carbon is present in large concentrations naturally as carbonate species in seawater, the environmentally significant carbon impact is the particulate fraction; its precision is estimated at 34 %. Both the dissolved and particulate nitrogen fractions are potentially important, but dissolved nitrogen is more significant for far-field eutrophication impacts. It accounts for ~78 % of the total nitrogen waste, and has a precision of 15 %. Note that the precision of the total discharges (dissolved + particulate) are closer to the better precisions for the dissolved fractions, because they are independent of the uncertainties in $f_{wastefeed}$ and f_{faeces}. The biggest contributors to the uncertainties in wastes expressed per tonne of fish produced are the metabolic (respiration or urine) / retained ratios, the FCR, and the composition of feed. None of the parameters that go into the growth model (TGC, Initial smolt weight etc) have much impact, because the normalization to fish produced removes such influences.

For model outputs expressed as the maximum discharges per day in the second calendar year of the grow-out cycle, the uncertainties range from 22 – 56 % (Table A.3) and are all higher than the corresponding ones for the wastes per tonne of production. This is because the maximum one-day discharge depends on fish size which in turn depends on water temperature, and is influenced most strongly by uncertainties in the TGC value. The other significant contributors to the uncertainty are the metabolic / retained ratios for C and N, and the feed composition. However, the other patterns for wastes expressed this way are similar to those for the per tonne production, with precisions of dissolved wastes (22 – 23 %) better than the precisions of particulate wastes (31 – 56 %), and the precision of dissolved N about twice as good as that for particulate C.

Table A.1. Values and estimated uncertainties (1 σ) for model input parameters used in calculations for SWNB.

Parameter	Units	Value	Uncertainty (1 σ)
TGC	$g^{1/3} \cdot {}^{o}C^{-1} \cdot d^{-1}$	0.00265	0.00022
Initial smolt weight	g	92	18
Julian day, start of grow-out cycle	d	121	27
Length of grow-out cycle	d	639	54
FCR	none	1.1	0.045
fraction of water in feed	none	0.1	0.045
f_{prot}	none	0.45	0.067
f_{fat}	none	0.26	0.045
$f_{carbohydrate}$	none	0.19	0.045
f_{ash}	none	0.096	0.013
fraction of P in feed	none	0.012	0.0013
fraction dry matter in whole fish	none	0.32	0.054
fraction C in dry fish	none	0.54	0.036
fraction N in whole fish	none	0.0288	0.0022
fraction P in whole fish	none	0.0045	0.00030
$C_{respiration} / C_{retained}$	none	0.87	0.27
$C_{faeces} / C_{retained}$	none	0.43	0.11
$N_{urine} / N_{retained}$	none	0.84	0.21
$N_{faeces} / N_{retained}$	none	0.28	0.058
$P_{urine} / P_{retained}$	none	0.6	0.13
$P_{faeces} / P_{retained}$	none	0.89	0.13
labile fraction of waste feed	none	0.5	0.089
labile fraction of faeces	none	0.5	0.089

Table A.2. Precision of model outputs (kg waste tonne^{-1} of fish produced) expressed as coefficients of variation (100 * σ / mean) determined from the Monte Carlo analysis. For each output, the three input parameters that contribute most to the uncertainty of the output are listed, together with the percentage change in the output (**Δ**) that corresponds to a 1 σ change in the input parameter.

Model output	**Value**	**CV (%)**	**Factor**	**Δ (%)**
$C_{dissolved}$	224	13	$C_{respiration}$ / $C_{retained}$	10
			Water in feed	5.2
			FCR	4.3
C_{part}	76	34	Water in fish	35
			$C_{respiration}$ / $C_{retained}$	30.1
			Water in feed	15.2
$O_{2\ demand}$	596	13	$C_{respiration}$ / $C_{retained}$	10
			Water in feed	5.2
			FCR	4.3
$N_{dissolved}$	33	15	f_{prot}	12
			N_{urine} / $N_{retained}$	9.1
			Water in feed	5.3
N_{part}	9.0	52	f_{prot}	43
			N_{urine} / $N_{retained}$	32.8
			N in fish	22.3
$P_{dissolved}$	4.95	13	P in feed	13
			f_{faeces}	7.1
			P_{urine} / $P_{retained}$	6.0
P_{part}	2.30	24	P in feed	28
			f_{faeces}	15.3
			P_{urine} / $P_{retained}$	12.9
$C_{dissolved} + C_{part}$	300	12	Water in fish	9
			Water in feed	7.8
			FCR	6.4
$N_{dissolved} + N_{part}$	42	19	f_{prot}	19
			Water in feed	8.3
			FCR	6.8
$P_{dissolved} + P_{part}$	7.2	13	P in feed	18
			Water in feed	8
			FCR	6.5

Table A.3. Precision of model outputs (maximum rate of discharge, calendar year 2, kg d^{-1} $(1000 \text{ fish})^{-1}$) expressed as coefficients of variation (100 * σ / mean) determined from the Monte Carlo analysis. For each output, the three input parameters that contribute most to the uncertainty of the output are listed, together with the percentage change in the output (**Δ**) that corresponds to a 1 σ change in the input parameter.

Model output	Value	CV (%)	Factor	Δ (%)
$C_{dissolved}$	4.5	22	TGC	20
			$C_{respiration}$ / $C_{retained}$	10.3
			Water in feed	5.2
C_{part}	1.53	40	Water in fish	35
			$C_{respiration}$ / $C_{retained}$	30.1
			TGC	20.2
$O_{2\ demand}$	12.0	22	TGC	20
			$C_{respiration}$ / $C_{retained}$	10.3
			Water in feed	5.2
$N_{dissolved}$	0.66	23	TGC	20
			f_{prot}	11.9
			N_{urine} / $N_{retained}$	9.1
N_{part}	0.181	56	f_{prot}	43
			N_{urine} / $N_{retained}$	32.8
			N in fish	22.3
$P_{dissolved}$	0.100	23	TGC	20
			P in feed	13.2
			f_{faeces}	7.1
P_{part}	0.046	31	P in feed	28
			TGC	20.2
			f_{faeces}	15.3
$C_{dissolved}$ + C_{part}	6.0	22	TGC	20
			Water in fish	9.4
			Water in feed	7.8
$N_{dissolved}$ + N_{part}	0.84	26	TGC	20
			f_{prot}	18.7
			Water in feed	8.3
$P_{dissolved}$ + P_{part}	0.146	23	TGC	20
			P in feed	18
			Water in feed	8.0