Waste outputs and dispersion around marine fish cages and the implications for modelling

Thesis submitted for the degree of

Doctor of Philosophy

By

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Dedicated

to

my family

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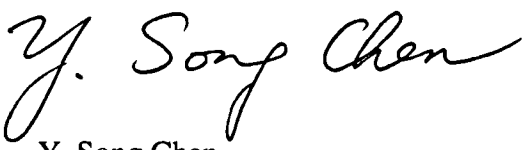
Good bye, Scotland, my second home, until we meet again.

DECLARATION

I declare that I carried out the work for and was principal contributor to the intellectual content of the papers published or in press in relation to this thesis (see Chapters for detail). No part of this work has been submitted for any other degree.



Y. Song Chen

ABSTRACT

Aquaculture plays an increasingly important role in food production as the catches of wild fish stocks continue to decline on a global scale through overfishing. However, the rapid development of intensive cage aquaculture in particular, which requires high inputs of energy, food and capital, can result in adverse effects on the environment. While spatial distribution and sediment loading models for particulate wastes from marine fish cages have been under development for more than 10 years, the models still contain numerous assumptions that limit their usefulness. These include the use of very limited data for fish feed and faecal pellets sinking rates that take no account of food manufacturer, type or size or environmental conditions. The present study provides information on a range of pellet types for three of the most important European farmed fish species (*i.e.* Atlantic salmon *Salmo salar* L., sea bream *Sparus aurata*, sea bass *Dicentrachus labrax*) that may be readily incorporated into models. Such data, combined with validation of predictions through *in situ* field investigations is designed to help improve the accuracy and usefulness of solid waste dispersal models.

The studies comprise four main sections, *i.e.*, quantifying food and faecal pellet characteristics, examining nutrient leaching rates from uneaten food and fish faeces, determining resuspension characteristics of uneaten food, modelling of solid wastes dispersion and thus the development of environmental tools. Existing literature relating to environmental impacts of cage aquaculture is reviewed and the key factors highlighted.

Two preliminary studies provided information on the influence of gravity acceleration on settling velocity determination and appropriate techniques for monitoring the rate of nutrient leaching from faecal wastes. Settling velocities of Atlantic salmon diets were significantly greater at 20 psu salinity than at 33 psu and significantly higher for most pellet types at 10 °C than at 20 °C. Settling velocities for unsoaked salmon diets were found to increase with pellet size, from a mean of 5.6 cm s^{-1} for the smallest pellet (2 mm) to 13.9 cm s^{-1} for the 10 mm standard (20 to 24% fat) pellets. Settling velocities of extruded diets for sea bream and sea bass diets ranged from 3.9 to 10.6 cm s^{-1}, broadly similar to those for salmonid diets. Settling velocities of salmon pellets were not significantly affected by immersion time (0 - 15 min). Given the water depths at fish cage sites and the settling times involved, it is concluded that it is unnecessary to take account of changes in food pellet settling velocity as a result of immersion. Freshly net-collected salmon faecal pellets appeared to consist of fine solid material approximately the size of the formulated diets. The range of salmon faecal settling velocities was 3.7 to 6.2 cm s^{-1} (mean = 5.3 cm s^{-1}) at 15 °C and 33 psu.

There are no significant differences in nutrient leaching of carbon and nitrogen from all six salmon diets after 20 min immersion in sea water. However, a rapid loss of faecal nutrients occurred 2.5 to 10 min after immersion in sea water. Total C and total N were found to leach by as much as 22% and 26%, respectively, after 5 min immersion during one sampling occasion.

Experiments conducted in a large-scale flume tank showed the critical resuspension velocities of a range of commercial fish feeds were between 8.63 cm s^{-1} and 9.53 cm s^{-1}.

Above the critical resuspension speed, pellets moved by saltation, *i.e.* traveling along the sediment by rolling, sliding or hopping on the bed. The velocities of pellet resettlement ranged between 0.79 cm $s^{-1}$and 3.98 cm s^{-1} under the critical resuspension speeds.

Field trials, involving the deployment of sedimentation traps, showed a general relationship between sedimentation of material and distance from cages, *i.e.* more sedimented material was associated with sampling sites closest to the cages. The spatial changes in sedimentation rates in the first trial were between 15.4 and 31.7 g DW m^{-2} d^{-1} at 30 m and 10 m stations, respectively. Values in the second trial (38.5-65.5 g DW m^{-2} d^{-1}) were twice those in the first trial, but followed the same pattern.

The model presented in this thesis is a combination of a spreadsheet model (Microsoft Excel 6.0) and Surfer plot program (Golden Software Ltd., ver. 6.04). Excel is used to prepare basic mathematical operations behind the model, including a mass balance submodel and use of a formula for calculating dispersion of uneaten food and faeces on the sea bed developed by Gowen *et al.* (1989). The operation of the waste dispersion model for marine cages takes into account the various settling characteristics of waste particles. It was verified with a set of *in situ* sedimentation data obtained from the field trial described above. Results described the waste dispersal around the vicinity of the cage farm.

For the future, it is intended that further validation and optimisation of the model will be carried forward by a combination of both increasing user involvement and incorporation of data from comprehensive studies as these become available. Together, these will contribute to reducing and remedying the environmental impacts of future development.

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Chapter 1

General Introduction

1.1 Cage aquaculture

Aquaculture is a rapidly developing global industry due to its profitability and the increasing demand for fish as population grows, as well as the near full exploitation of natural fisheries resources. Global fish production was 122 million tonnes in 1997 (FAO, 1999a). Capture fisheries production in 1997 remained stable at 93.3 million tonnes which accounted for 76% of the total, while aquaculture production increased by 7.6% to 28.8 million tonnes (FAO, 1999a, b). It has been often suggested that aquaculture will expand to compensate for shortfalls from catches. Especially a slowdown has been witnessed in the growth of fish supplies in recent years, and is likely to continue. Certainly, at the present trend, with aquaculture as the only fisheries sector to be increasing, would appear to support this contention. Hence, offshore areas beyond the immediate nearshore waters are increasingly beckoning to be exploited for cultivation (Stickney, 1997). Among the various aquaculture systems that are available, cages are probably the most versatile and can be deployed in inland waters, estuaries, coastal regions and the open sea. With improvements in technology, there has been diversification in the types and designs used. These can be broadly classified into four basic types: fixed, floating, submersible and submerged (Beveridge 1996). Over the past twenty years, there have been many developments in cage technology, especially in coastal and offshore marine designs. Due to the numerous advantages offered by cage aquaculture, including lower labour costs per unit production, their ease of management and access to space, and their higher holding capacities per unit area compared with land based systems, production of cage aquaculture has greatly increased in developed countries particularly with respect to the intensive cage culture of

marine fish (Huguenin, 1997). Most of this expansion has been with farmed salmonid species and still seems promising for the immediate future, 1999 being one of the best years for salmon farmer in living memory, except perhaps in Scotland and New Brunswick, Canada (http://www.intrafish.com). There is, nevertheless, considerable worldwide diversity in terms of cage culture species and culture conditions. Since the trends are toward larger cages and more exposed sites, there are many negative impacts of intensive cage aquaculture that must be considered if the industry is to continue to develop. The potential problems/impacts of cage aquaculture development and operations are discussed in next section (Section 1.2).

1.2 Environmental impacts of cage aquaculture

All aquaculture systems reliant on exogenous feed or other inputs have the potential to impact on the environment and the more resources that are used, the greater the quantities of wastes that are likely to arise. The caged fish are dependent on the environment to provide a wide range of environmental goods and services including the steady supply of dissolved oxygen and the removal and assimilation of wastes. The essential difference between cages and land-based aquaculture is the latter are usually located in areas of agricultural land (*i.e.* an enclosed system in already exploited ecosystems), whilst the former requires space in the sea (*i.e.* an open system normally in unexploited ecosystems). As cages are essentially ecologically open systems, there are inevitably wastes produced that are released into the surrounding environment. Unlike land-based aquaculture systems, such as ponds or tanks, where it is possible to introduce some type or other of waste

treatment before effluents are discharged into the environment, marine cage systems discharge the untreated wastes directly into the environment. In intensive aquaculture, nutrient enrichment primarily results from uneaten food and fish faeces (Phillips *et al.*, 1985; Lumb, 1989; Ackefors and Enell, 1990, 1994; Blyth *et al.*, 1993; Henderson *et al.*, 1995; Beveridge, 1996; Black *et al.*, 1996). Since the 1980s, there has been increasing concern about the environmental impacts of cage aquaculture, especially the wastes from its operation and production processes. Wastes from cage systems primarily consist of uneaten food, metabolic waste (faeces and urine) and chemical wastes (Beveridge *et al.*, 1991; Beveridge, 1996). The bulk of the wastes are solids and are thus subjected to sedimentation. The sedimentation of the solids is dependent upon on their settling velocity which in turn is dependent upon pellet shape and density, current velocity, turbulence and depth of the sites (Hansen *et al.*, 1991; Silvert, 1992; Gowen *et al.*, 1994; Elberizon and Kelly, 1998; Chen *et al.*, 1999 a, b). In general, the impacts occur on several space and time scales, which can be classified as internal, local, and regional (Silvert, 1992; Beveridge, 1996). Internal impacts are those involving the effects of a particular farm on itself (*e.g.* fish within the cages) and its immediate environment, generally on a scale of a few hundred meters and fluctuations over times measured in minutes or even seconds (Silvert, 1992; Sowles *et al.*, 1994). However, some internal impacts such as those associated with fouling, can operate over much longer time scales. Local impacts can affect nearby farms and wild populations over distances in the order of a kilometer (Silvert, 1992). Regional impacts involve an entire inlet or larger water body (*e.g.* a whole bay or fjord) with space scales of many kilometers and time scales ranging from a single tidal cycle to an entire season. One example of internal impact is oxygen depletion by the fish within the boundaries of a single fish cage.

An obvious form of local impact is the deposition of uneaten feed and faecal matter on the seabed under the cages, and this has received widespread attention in the literature (Gowen and Bradbury, 1987; Waldichuk, 1987; Hargrave *et al.*, 1993; Johannessen *et al.*, 1994; Tsutsumi, 1995; Findlay *et al.*, 1995; Beveridge, 1996; Black *et al.*, 1996; Hevia *et al.*, 1996; Hargrave *et al.*, 1997; Yokoyama *et al.*, 1997; Wu *et al.*, 1999). Most studies of the environmental impacts of cage aquaculture have shown increases in the levels of suspended solids and nutrients (ammonia, organic nitrogen and carbon) which result in eutrophication (Ackefors and Enell, 1990; Angel *et al.*, 1992; Holmer and Kristensen, 1992; Wu *et al.*, 1994; Silvert and Sowles, 1996, Black *et al.*, 1996), and decreases in dissolved oxygen around the cages (Brown *et al.*, 1987; Holmer and Kristensen, 1992; Tsutsumi, 1995; Silvert and Sowles, 1996). There is considerable increase in oxygen consumption and the accumulation of total nitrogen, total carbon from solid wastes, especially in the sediments immediately below cages. High waste deposition rates can cause an accumulation of organic detritus in the sediments that overwhelm the assimilative capacity of the benthic community and result in the formation of anaerobic bacterial mats and anoxic conditions. This leads in turn to the generation and out-gassing of hydrogen sulphide and methane. These effects are well recognized because they affect both the farms and the environment, with visible changes to the benthos and recognizable changes in wild benthic populations in the vicinity of fish farms (Phillips and Beveridge, 1986; Brown *et al.*, 1987; Holmer and Kristensen, 1992; Black *et al.*, 1996).

Cage farming in sheltered bays, fjords and other coastal areas involves the use of highly intensive stocking and feeding which results in increased sedimentation, biochemical

oxygen demand and nutrient loadings. The crowding of fish cages in semi-enclosed or near-shore areas has frequently resulted in self-pollution and transmission of diseases (*e.g.* Wu *et al.*, 1994; Findlay *et al.*, 1995). There have also been conflicts with other users of the coastal areas, such as recreation, navigation and those who enjoy the scenic beauty of waterfront areas, forcing regulatory agencies to control the expansion of these aquaculture developments (Sien, 1979; Wohlwill, 1982; Beveridge, 1984, 1996; O'Sullivan, 1991; Kryvi, 1995; Wu, 1995).

Sustainability is a concept that refers to management practices based on the prudent use of renewable and/or recyclable resources that will not degrade the environment, so that natural resources are continually regenerated (Chamberlain and Rosenthal, 1995; Chua, 1997). It is obvious that a sustainable use of the environment has to be one that does not bring about irreparable harm to the ecosystem, but limits the inevitable changes within the boundaries of natural fluctuation. Appropriate control has to be enforced to ensure such rational use, but to be effective, guideline and regulations have to be based on adequate knowledge of the impacts of developmental alternatives and possible means of mitigating the adverse effects of selected uses (Pillay, 1992).

1.3 The present study

A focus of research here is modelling the nutrient loadings from cage aquaculture in order to predict impacts to allow more effective environmental management. In general, there are two components to modelling waste dispersion from cage aquaculture; (1) determination of

the quantity and quality (*i.e.* form) of wastes entering the environment, and (2) determination of where the waste is going, and in what quantity, in terms of two or three-dimensional coordinates. To quantify waste entering the environment from uneaten food and faecal material either simple mass balance models or direct measurement can be employed. However, at present, models to calculate dispersion of solid wastes in the marine environment rely on scant data sets or assumptions on settling velocities and dispersal behaviour of food and fish faeces that take no account of particle size, environmental conditions (*e.g.* temperature, salinity), feed formulation and the re-suspension and degradation of settled food and faecal pellets. Given the above information and the lack of comprehensive studies on the relationship between the physical characteristics of feed/faecal pellets and their settling velocity, it was decided that the present study would address these deficiencies. The present research would also focus on modelling the nutrient loadings of cage aquaculture, to predict the environmental impacts of cage aquaculture so that we may better match environmental capacity with production. The thesis includes four main sections: the quantification of food pellet characteristics, examination of nutrient leaching rates from uneaten food and fish faeces, investigation of critical re-suspension speed of feed pellets, modelling of solid wastes dispersion and development of environmental modeling tools. Specific objectives were:

1. To determine relevant physical characteristics and settling velocities of a range of commercial feeds under defined laboratory conditions. Waste dispersion models applied to the cage Atlantic salmon (*Salmo salar* L.) industry in Europe continue to incorporate the waste feed settling values provided by Gowen and Bradbury (1987), which give no indication of pellet size. Not only might feed settlement rates be expected to vary with

pellet size, but also perhaps with environmental conditions (temperature, salinity) and with formulation.

2. To carry out chemical analysis of salmon diets and faeces in order to provide the data for determining the nutrient loadings at Atlantic salmon sea cage farms. It will result in a better understanding of the mass balance of nutrients, carbon (C) and nitrogen (N), in marine cage farming industry.
3. To conduct laboratory-based leaching experiments and to find out the leaching rate of C & N from both diet and faecal pellets during certain time period and to predict the amount of solid waste leached as they sink through the water column.
4. To conduct laboratory-based re-suspension experiments and to determine the critical re-suspension speed of feed pellets, also the resettlement velocity of feed pellets under those critical re-suspension velocities.
5. To carry out a field study around a cage farm by employing sediment traps and determine how much of the nutrients originating from farm sites by measuring accumulation *in situ* in sediment traps.
6. To create a model with low data requirements. The model should also be user-friendly and easy to run by the regulatory authorities. To update the derived leaching rates to the nutrient balance in a marine cage farm as well as to refine the existing model for prediction the fate of nutrients in marine cage farms. The ultimate aim is to provide a map that is sufficiently fast and accurate for management purposes.
7. Verify the model by combining modelling and field work (sedimentation rate with dispersion model). This may prove useful in assessing the fate of wastes (dispersion

area and amount) and make it possible to determine the optimal stocking density in situations where intensive field work is not possible.

The thesis comprises seven chapters and nine appendices. Chapter 2 reviews and defines modeling and outlines potential applications of modeling to cage aquaculture waste management as well as the criteria that constrain the types of model that are suitable for solid wastes dispersion. This chapter also presents a conceptual model of the processes that require consideration in the development of practical models. Chapter 3 and 4 investigate respectively the principal physical and chemical characteristics of the feed and faecal pellets. The settling velocity and nutrient leaching rate of feed and faecal pellets of immediate importance to the operation of the waste dispersion model are determined here. Chapter 5 presents work on the determination of critical re-suspension velocities of the feed pellets under defined hydraulic laboratory conditions and the field work at the cage farm for determining the quantity of solid wastes designed to subsequently validate the modeling work in Chapter 6. This chapter 6 provides practical farm production data for incorporation into the revised model, from simple to complex, providing contour maps of wastes dispersion to deal with real and potential waste management issues. Finally, Chapter 7 presents a general discussion and the implications for carrying capacity for certain marine situations, especially Taiwanese cage aquaculture, and gives some insights into its development. It also draws conclusions from the study and makes recommendations for future study.

Chapter 2

A review of some aspects of aquaculture modelling and monitoring of environmental impacts

2.1 Environmental impacts assessment and modelling

Intensive aquaculture can release considerable amounts of wastes to the environment, it needs to be controlled and regulated. Aquaculture development is similar to other developments and the term 'impact' and 'effect' are frequently used synonymously. An 'impact' describes changes in an environmental parameter over a specified period and within a defined area resulting from a particular activity (Wiesner, 1995). These changes are compared with the previous, baseline situation which existed before the activity had been initiated. Since Green (1979) proposed the Before-After/Control-Impact (BACI) design of monitoring program to detect environmental impacts, the concept has become considerably more sophisticated, powerful and sensitive. The normal procedure in assessing the changes and the degree of impact produced by aquaculture or other activities is to carry out an environmental impact assessment (EIA). The EIA can be defined as a procedure for evaluating the environmental impacts of a product, process or activity (Morrisey, 1993; Priyan and Smith, 1994; Gilpin, 1995; Wiesner, 1995). There are at least three interpretations of the term 'environmental impact assessment' (Lincoln-Smith, 1991). *A priori*, it can refer to prediction of changes in natural entities in response to a disturbance, or to prediction of the cost (economic, social, environmental, aesthetic, *etc.*) resulting from those changes. In addition to those predictions, EIA can also involve measuring the changes that actually occur when the disturbance takes place. The third interpretation combines the first two into a formal process in which predictions are framed as testable hypotheses and the monitoring of subsequent changes provides the test (Hilborn and Walters, 1981; Peterman, 1990; Lincoln-Smith, 1991). It is generally accepted however, that EIA should

embrace the evaluation of social, economic and ecological impact of a proposed development as well as the identification of impact mitigation measures and alternative development options (GESAMP, 1991b). From an ecological perspective, the purpose of an EIA must be to predict and assess the ecological consequences of the planned facility, thereby providing the regulatory authority with the scientific basis to determine the acceptability of the perceived impacts relative to pre-determined standards. Hence, the use of EIA in the management of marine aquaculture development requires the application of ecological knowledge. However, EIAs have frequently consisted of collections of largely descriptive data, with little *a priori* consideration of the specific changes to be expected from the proposed development (GESAMP, 1996). It is therefore suggested that specific, testable hypotheses be formulated at an early stage in the EIA process (Jorgensen, 1991a). Based on this, there are many tools can be used such as checklists, environmental quality indices, overlay mapping, network diagrams, system diagrams, simulation models (Jorgensen, 1991a), the latter being a particularly cost-effective tool (Silvert, 1992; Leung and El-Gayar, 1997). Modeling is also the most advanced and the most quantitative EIA method available (Jorgensen, 1991b).

A model is a simplified representation of reality. By 'simplified', it is meant that a model contains only those attributes that are relevant to a particular problem (Leung and El-Gayar, 1997). It can be a collection of mathematical equations or rules that describe fundamental relationships between components of a system. Therefore, a model provides a rigorous framework for characterizing important variables, relationships, and processes in a system. In general, equations represent relationships that are quantifiable, while rules represent relationships that are at least partly qualitative in nature (Piedrahita *et al.*, 1999). Models

are useful tools for representing and experimenting with complex reality. Modelling thus refers to the act of constructing, validating, verifying, and using models. Why is there a need to model aquaculture systems? Cuenco (1989) provides the following answers to the question:

1. Modelling serves as a powerful tool for the formulation, examination, and improvement of hypothesis and theories.
2. Models can make intelligent predictions about the consequences of various management strategies on the system.
3. Modelling provides a working tool to conduct numerous 'what if' experiments quickly and to evaluate the consequences of various hypothesis or management strategies of large and complex aquaculture system, which are seldom possible in their natural environment.
4. Models facilitate the evaluation of complex interactions of aquaculture systems.
5. Modelling accelerates the use of more quantitative and precise methods in aquaculture research.
6. Models put together knowledge from theoretical, laboratory, and field studies into a consistent whole so as to identify areas where knowledge is lacking, sparse, or inconsistent.

In general, a model is used because of its economy, availability, and ease of information handling. The purpose of modelling can be briefly divided into three levels, *i.e.* (i) descriptive, (ii) monitoring, (iii) explanation and prediction. It may cost less to derive knowledge from a model than from the real world counterpart such as large, complex aquaculture system. A model may represent an aquaculture system that does not exist or

cannot be easily manipulated. In addition, a model may provide a convenient medium to collect and transmit information. Models of aquaculture systems are largely extensions of models of agriculture system, with special attention to the handling of the growing medium (Leung and El-Gayar, 1997).

Sustainability (Section 1.2) as it refers to aquaculture can be defined as a group of continuously evolving aquacultural production and marketing systems that are sustainable in the long term for all users and consumers (Chua, 1997). The current drive towards sustainable aquaculture is expected to emphasize further the need to model the environmental impacts of aquaculture. Of particular importance are models that seek to improve water quality and feed use as well as to decrease the environmental impacts of aquaculture. Modelling is an alternative tool for predicting and evaluating the development of such management practices. Nevertheless, model development is a slow and time-consuming (*i.e.* sometimes expensive) practice and efforts therefore normally have focused on research models (Leung and El-Gayar, 1997). Eventually, aquaculture models should be developed to the point where they become important tools for designing and managing aquaculture systems. Using a model to evaluate the success or failure of a proposed system under various conditions is much cheaper than building the system itself and then trying to sort the problems. Thus, costly trial and error methods can be avoided (Cuenco, 1989).

2.2 Model types

2.2.1.Classification of models

There are numerous models and the approaches to modelling, and some of the following distinctions may be made in describing the model type. In general, models can be classified by how they have been constructed: 1) empirical models, *i.e.* usually derived from large data-sets, 2) mechanistic (theoretical) models, *i.e.* from theory-deducing and 3) empirical-theoretical models which combine the two. It is outwith the scope of this brief introduction to include all possible classes of models so discussion is confined to those most likely to be encountered. It is often useful to distinguish between various pairs of model types, as following distinctions shown in Table 2.1. Hence, models are normally segregated into stochastic and deterministic models, dynamic and static models, lumped parameter and distributed parameter models, linear and non-linear models, research and management models (see Jeffers, 1978a; Jorgensen, 1983; GESAMP, 1991a).

Empirical-Mechanistic (Theoretical) models

The two most general types of models are distinguished as empirical models and mechanistic (theoretical) models. An empirical model is developed primarily from an analysis of data, fitting equations to the data rather than basing them on more theoretical principles. Empirical models set out principally to describe situations in which the ecosystem or culture environment is treated as a 'black box' with only inputs and outputs (Figure 2.1). This means that predictions cannot be used to determine conditions outside those encountered at the time of data collection. Processes that take place inside the system are also ignored. Mechanistic models on the other hand, are intended to be mathematical

descriptions of theoretical principles. They attempt to provide an understanding of the biological and environmental processes of the ecosystem or culture environment on a finer scale than empirical models. Mechanistic models are complex syntheses of what is known of the ecosystem and hence are very difficult to build and use. They are sometimes referred to as 'internally descriptive' or 'theoretical' models. However, those two categories are not mutually exclusive, and a good model may contain both empirical and mechanistic features.

Deterministic-Stochastic models

A deterministic model uses single expected values for all parameters and variables, and produces single expected yield predictions, for which the predicted values may be computed exactly. Stochastic models incorporate variability, and possibly error, by using probability density functions for selected parameters and/or variables. This results in a probability density function for the prediction, for which the predicted values depend on probability distributions (Jeffers, 1978b).

Static-Dynamic models

This refers to the presence or absence of time-dependency. Static or steady-state models describe behaviour that is constant over time and thus are time-independent. Dynamic models describe behaviour that varies with time and thus are time-dependent.

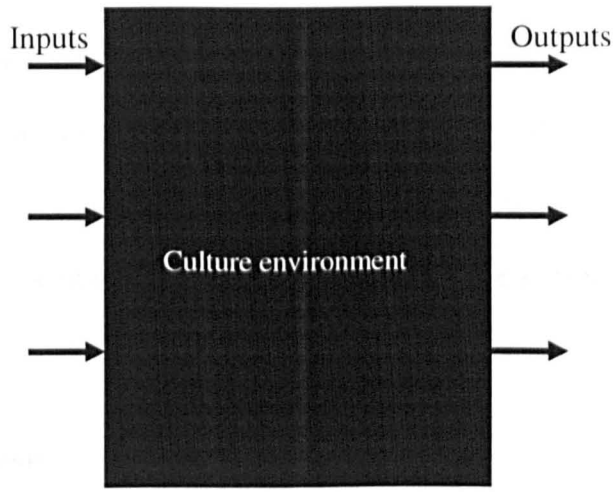
Lumped parameter-Distributed parameter models

This refers to the presence or absence of a space-dependency. Lumped parameter models are zero dimensional in space; they are based on an assumption of uniform conditions throughout the system modeled. Distributed parameter models, on the other hand, are developed to describe systems with variable conditions in one or more spatial dimensions.

Table 2.1 Classification of models (pairs of model types, modified from Jorgensen, 1983 and GESAMP, 1991a)

Model Type	Characteristics
Empirical models	Derived from large data-sets
Mechanistic models	Derived from theory-deducing
Deterministic models	The predicted values are computed exactly
Stochastic models	The predicted values depend on probability distribution
Static models	The variables defining the system are not dependent on time
Dynamic models	The variables defining the system are a function of time (or perhaps space)
Lumped models	The parameters are within certain prescribed spatial locations and/or time, considered as constants
Distributed models	The parameters are considered functions of time and space
Linear models	First-order equations are consecutively used
Non-linear models	One or more of the equations are not of the first order
Research models	Used as a research tool
Management models	Used as a management tool

(a) Empirical Model



(b) Mechanistic Model

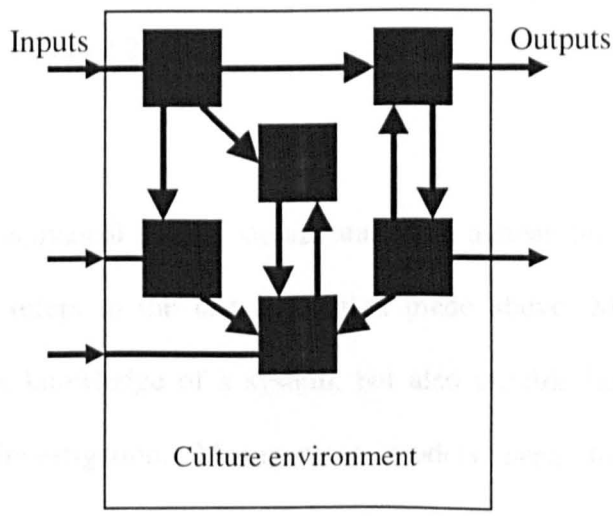


Figure 2.1 Schematic illustration of the two types of biological models. (Reproduced from Leung and El-Gayar 1997).

Nonlinear-Linear models

In general, lumped parameter models are nonlinear. However, the modeller will generally strive to obtain a linear model, since this may be much more comprehensively analysed, and obeys the principle of superposition.

Research-Management models

Most models discussed in this thesis are management tools, although almost all of the models also can be used in the research context to find new relations or to identify research needs. However, the purpose of model development will to some extent determine its form.

2.2.2 Model development

The primary focus of model development is to define the goals and objectives. Only in this way can it be ensured that limited research resources can be correctly allocated and not dispersed into irrelevant activities (Jorgensen, 1983). A tentative guideline of modeling procedure is presented in Figure 2.2.

Goals and Objectives

The development of mathematical models should start with a clear definition of goals and objectives. This mainly refers to the last distinction made above. Models for research purposes aim to improve knowledge of a system, but also provide indicators for further fruitful directions of investigation. Management models need to be directly and immediately applicable, preferably with a minimum data requirement.

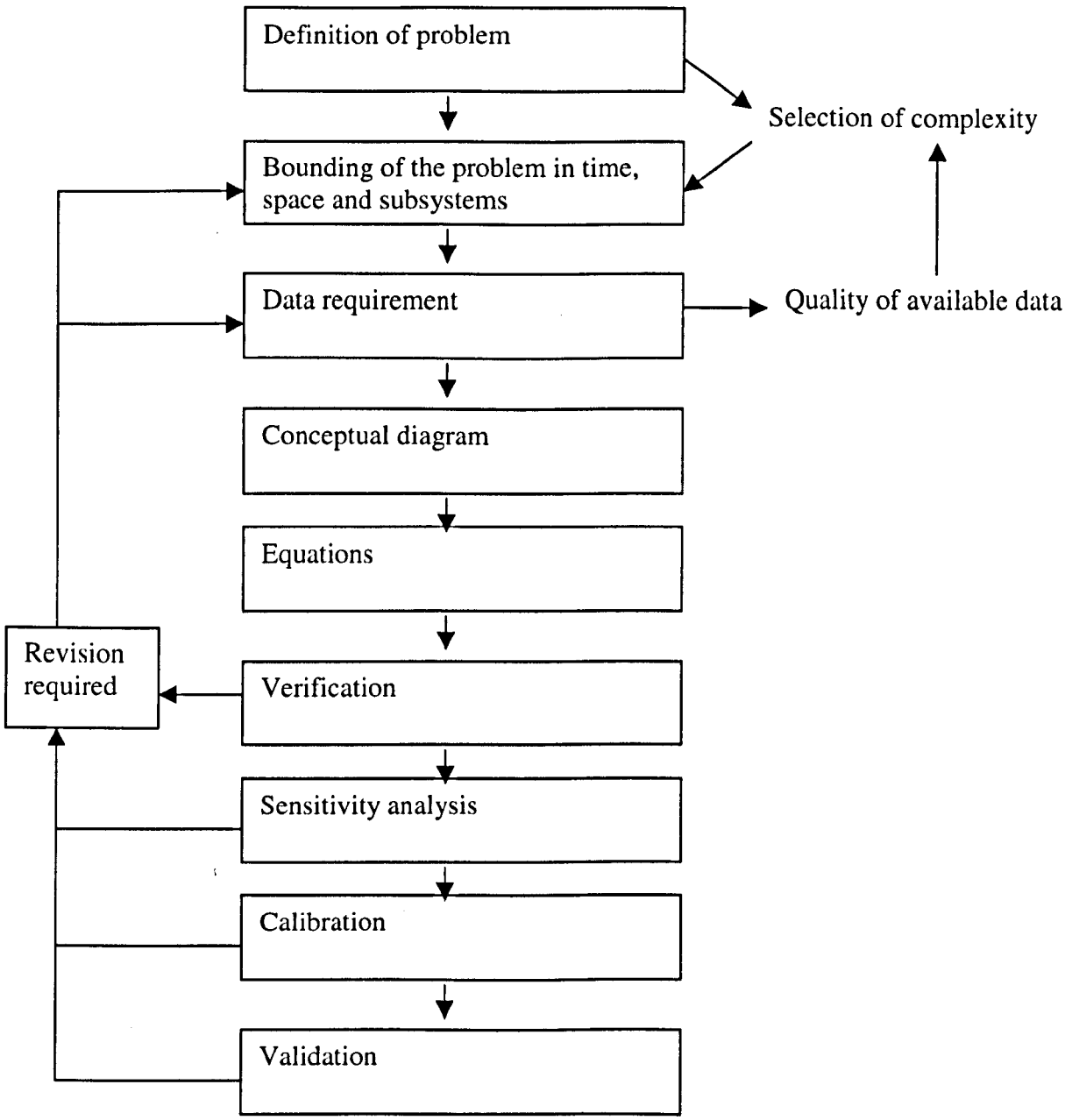


Figure 2.2 A tentative guideline of the model development

Conceptualisation

With objectives defined, the next step is conceptualisation. This is the description of the system, with all the major factors defined. This usually involves a review of existing knowledge of the system that is to be modeled. It almost inevitably requires some grouping of factors into single functional or spatial groups, i.e. all the phytoplankton species, or all the cage discharges. There is no guarantee that the choices made here will turn out to be correct at later stages of the modelling procedure. The dismantling of a complex environmental system for individual analysis of component parts does not always mean that subsequent reassembly will yield a characterisation of the behaviour of the whole. The model type will be selected, and the possible paths of interaction between variables identified.

Formulation

Once the main factors of the model have been defined, formulation step seeks to identify the relevant variables and define expressions to describe their interactions. Different types of variables may be identified as follows (Beck, 1983),

- Measured input disturbances: these are the input variables to the system, e.g., suspended solids loadings, or nitrogen input from a fish farm, which can be measured and quantified.
- Unmeasured (unknown) input disturbances: these are unmeasured input variables such as diffuse non-point carbon or nitrogen loading to the nearby cage. A predominant characteristic is that they generally exhibit random variability.

- Process state variables: these are the internal variables of mechanistic models that characterise the essential properties and behaviour of the system, as functions of time and space.
- Parameters: these are functions of other variables, usually rate coefficients, which should be as constant and independent of space and time as possible.
- Measured output variables: these are often simply measurements of some of the process state variables. However, they may be more complex, *e.g.* the sum of several state variables to give an output variable, such as total carbon and nitrogen.
- Measurement errors: these derive from process instrumentation and laboratory analysis. Such errors are inherent in all measurements and thus the measured output variables are never an exact measurement of the process state variables.

Analysis

Formal analysis is the confrontation of the model with actual data and includes the following functions: observation and measurement, sampling and experimentation. It is important that these are carried out with regard to conventional scientific and statistical methodology, and that associated error is taken into account.

Solution

This includes computer implementation, and the selection of specific solution procedures. The data are analysed and the model used to give results. The form of the results will depend on the type of model employed.

Testing and evaluation of model

A major issue is uncertainty, or conversely, reliability. Various techniques for evaluating this are available and include those of first-order analysis, and Monte-Carlo simulation. Various forms of sensitivity analysis may also be used.

Beck (1983) makes the distinction between verification and validation: the former is the determination of whether the 'correct' model has been obtained from a given single set of experimental data. Validation requires that a second independent set of field data is used to further test model applicability. Nevertheless, he recognizes the true philosophic sense of the word, no model can ever be completely validated. For this reason, Reckhow and Chapra (1983) prefer to speak of model confirmation and corroboration.

This procedure is not necessarily rigid, and new information and results are constantly used to update and improve model structure. The following section provides an overview of aquaculture modelling with particular emphasis on management applications such as waste dispersion modelling and carrying capacity modelling.

2.3 An overview of current research of aquaculture modelling in monitoring environmental impacts

Current models that are used in aquaculture management, are mainly for predicting eutrophication (*e.g.* Schnoor and O'Connor, 1980; Rossi *et al.*, 1986; Meyers *et al.*, 1999) and carrying capacity studies (*e.g.* Newell, 1988; Paller, 1992; Raillard and Menesguen, 1994). The latter type of model, in particular, has developed rapidly during the past decade due to increasing concerns about the environmental impacts resulting from increasing development of aquaculture. A considerable amount of research is targeted at providing a scientifically based method to predict or determine the environmental impacts of anthropogenic nutrient sources such as crude sewage, settled and treated effluent discharged into the estuarine and marine environment. Although the SEPA BenOss model, for example, was developed to meet an immediate need arising from specific issues, it is applicable to any study of other organic matter discharged to the sea (Cromey *et al.*, 1997). Environmental consultancies working in the aquaculture sectors will have details of analysis of environmental impact of sea cage farms as well as internal management programs designed to ensure compliance with environmental standards. Consultants will acquire only the environmental data necessary to review a consent and to assess sites for development or expansion. Data collection and analysis procedures based on legal standards are consistent and therefore allow comparisons between different sites and times. While models developed for aquaculture can be classified in several ways based on the specific model characteristics chosen for classification, it is instructive to broadly categorize aquaculture models as biological, economic, biophysical, and bioeconomic, as this emphasizes and clarifies the underlying approach for which the model is built (Leung

and El-Gayar, 1997). As stated in Section 2.2.1, the concept of an ecological model could be interpreted as a mathematical representation of a natural process or processes. However, it is impossible to incorporate all the factors involved and the natural processes and interactions into one ecological model because of the lack of knowledge of complex natural fluxes. Therefore when using a model only the main variables (*e.g.* carbon and nitrogen loading from cage farm) which govern a process are defined and represented in a mathematical way.

Biological models are primarily concerned with modeling the biological processes involved in aquaculture-related activities. Such models are generally difficult to construct because of the complexity of the biological organisms and their interactions with the environment. In many cases, particularly in pond culture, the aquaculture manager or fish farmer has very little knowledge of how the production in the pond is progressing (Leung and El-Gayar, 1997). Unlike traditional agricultural or terrestrial livestock production, the aquaculture producer cannot directly visualize the growth of the 'crop' and therefore must rely on indirect and subjective measure of production on which to base management decision. There are three crucial factors in pond aquaculture production that are not essential for other animal production systems, namely, ecology, monitoring of animals, and feed utilization, and these also require additional modeling sophistication (Hatch and Kinnucan, 1993). However, cage systems, with modern technology and design, are much easier to obtain production data from than ponds. For example, the exact feeding rate and intake rate and thus uneaten feed ratio, or fish biomass estimation from the cage farm are readily available from present day feeding systems. Nevertheless, the cage is still widely regarded

as a black box. In the present study, it is aimed at unveiling some of the contents of this black box – in a sense making it somewhat 'translucent'.

Numerical models have the potential to provide qualitative predictions and are therefore a useful tool in quantifying impacts resulting from aquaculture waste. A combination of field investigation and modeling is recommended since this will introduce scientific rigour into the evaluation process (GESAMP, 1996). Thus, the outputs from models might be used to develop hypotheses regarding the extent of ecological change that are tested using data collected during the monitoring program. The use of models may also be cost-effective by reducing the amount of field investigation required, and may help to illustrate impacts in ways that are more readily interpreted. Thus, it is essential to show an example of using predictive models to determine the level of exploitation of a particular site and the scale of monitoring that should be undertaken.

There are many types of models that can be applied to aquaculture processes and wastes. Some of these models deal with nutrients, chemicals, solid wastes, etc. The nutrient models are mainly based on calculating phosphorus (Dillon and Rigler, 1974; Beveridge, 1984; Foy, 1992; Ackefors and Enell, 1994; Kelly, 1995; Hakanson *et al.*, 1998) and nitrogen (Turrell and Munro, 1988; Gowen *et al.*, 1992; Ackefors and Enell, 1994; Troell and Norberg, 1998) losses to the environment. Chemical dispersion models have been developed for individual compounds because of their different routes and behaviours in the environment. These are likely to play an increasingly important role as the demand for chemical applications is increasing world-wide (*e.g.* Falconer and Hartnett, 1993; Burns *et al.*, 1999). Solid wastes dispersion models are based on dispersion equations, *i.e.* predicting

where a particle released from a farm will settle on the seabed (Gowen *et al.*, 1989). The model prediction is compared to an Environmental Quality Standard (EQS) so that the degree of environmental exploitation (maximum size of farm) can be decided. The results of the monitoring program are used to assess the EQS, evaluate model predictions and ensure that the current operation or the degree of exploitation does not compromise the EQS. However, in most cases, an EIA has not been undertaken prior to development of an aquaculture operation. Hence, aquaculture monitoring programs often fail to fulfill their function because of the absence of a management framework with pre-determined standards (Beveridge, 1996; GESAMP, 1996). On occasion, monitoring has been imposed as a result of public pressure because of the perceived ecological damage caused by aquaculture wastes, often resulting in the measurement of a wide range of ecological variables, many of which are inappropriate, or the collection of data that are difficult to interpret, or failure to analyze and interpret data and implement feedback mechanisms to modify farm production and the monitoring program itself (GESAMP, 1996). Even where monitoring has been undertaken, when socio-economic and political pressures take precedence over ecological concerns, due to budgetary, manpower and organizational constraints, the implementation of such monitoring programs may still have been inadequate (GESAMP, 1996).

2.4 Nutrient loading models (mass balance models)

Quantifying the amount and type of waste generated from fish farms is the first step in predicting the scale of effect (Gowen, 1994). Mass balance models for methods to predict

environmental effects and to quantify the output of waste have been presented for different types of ecosystems (*e.g.* Penczak *et al.*, 1982; Braaten *et al.*, 1983; Beveridge, 1984; Gowen and Bradbury, 1987; Aure and Stigebrandt, 1990; Ackefors and Enell, 1990, 1994; Pridmore and Rutherford, 1992; Troell and Norberg, 1998). However, system characteristics and management are crucial in determining in what form and when wastes are released. Moreover, mass balance models give no information on the subsequent fate of farm effluents in the environment (Beveridge, 1996). Attempts to model hypernutrification and eutrophication at present appear to be considerably less successful in coastal waters than in inland waters. The same basic principles of relating nutrient concentration to phytoplankton growth, together with a dilution or water-body flushing time, may apply (Barg, 1992). However, difficulties in modelling coastal ecosystem responses to nutrient enrichment are generally related to the influence of salinity stratification and tidal mixing, particularly in embayments and estuaries. The boundaries of the affected area are usually difficult to define. As a consequence of the complex linkages between biological, chemical and physical process, the models are area-specific and have limited wider use (Barg, 1992). The extent of hypernutrification is associated with the size of the farm and hydrography of the water body within which the farm is located. Hence, the water body volume, its exchange rate with the adjacent sea, the onset and duration of vertical stratification and the extent of horizontal advection, all make important contributions to the levels of hypernutrification. An example, assuming complete dispersion of waste nitrogen throughout a semi-enclosed water body, is presented in the following approach, to estimate the equilibrium increase of soluble nitrogen concentration by Gowen *et al.* (1989) as follows

$$Ec = \frac{N \times F}{V} \qquad (2.1)$$

Where:

E_c is equilibrium rise in concentration (level of hypernutrification) (m mol. m^{-3}),

N is daily output of soluble nitrogenous waste (m mol. d^{-1}),

F is flushing time of the water body in days (d),

V is volume of the water body (m^3).

The initial work on nutrient loading models was based on input-output mass balance equation, *i.e.*

Loading = Input - Output ± Transformation

Mass balance calculation can indicate the amounts of carbon and nitrogen released from uneaten food and fish faeces in Atlantic salmon cage aquaculture. Estimates of the area of wastes dispersion can be derived form the mass balance model in conjunction with hydrography data. A mathematical model, originally described by Gowen *et al.* (1988) for estimating loadings from marine cages, has been under a continuous process of development by staff at the Institute of Aquaculture and applied for some years (Institute of Aquaculture, University of Stirling). The model was based on a complex multi-layered mass balance spreadsheet model (data stored in Quattro Pro®). Although it accounts for carbon input from feed and faeces and also attempts to predict the overall displacement of carbon loadings distributed in the sediments, it retains numerous assumptions such as the use of a single FCR value, and that no resuspension and nutrient leaching occurred after sedimentation which without taking account of site-specific environmental conditions (*e.g.* temperature, salinity), make it somewhat unrealistic. Thus, in the present study, it attempts

to derive more data and incorporate those realistic parameter values and conditions for more effective wastes dispersion modelling. Moreover, the model has not been verified with field data. Thus will also be addressed in the present study.

2.5 Waste dispersion models

The most frequently reported and best characterised of the effects of marine cage aquaculture are on the benthic environment due to the rapid settling velocities of solid wastes, especially uneaten food pellets under or near to the cages. Hence, waste dispersion or sedimentation models have been developed to predict the magnitude and spatial extent of particulate matter deposition. These models typically attempt to predict the trajectory of particulate wastes based on the hydrographic regime and settling velocities of feed or faecal matter (Al-Rabeh and Gunay, 1992; Falconer and Hartnett, 1993; Kishi *et al.*, 1994; Silvert, 1994; GESAMP, 1996; Panchang *et al.*, 1997; Cromey *et al.*, 2000). As stated in GESAMP (1996), there are five potential applications for dispersion (sedimentation) models:

(i) Site selection: Given a particulate production level, the model would be used to establish the deposition rate of particulate wastes at alternative sites with reference to establish environmental quality standards.

(ii) Defining site limitations: In the converse application of the same models, it is possible to establish the maximum production attainable at a site, given a maximum permissible loading of particulate matter.

(iii) Determining responsibility: There is a variety of natural and anthropogenic sources of organic material introduced to aquatic environments. A sedimentation model could be used

to determine the relative contribution from each source provided adverse ecological consequences of enrichment are observed.

(iv) Optimising production: Models can be used as indications of the extent to which enrichment has a deleterious effect on production.

(v) Design of monitoring programs: The magnitude of predicted carbon loadings can be used to establish an appropriate intensity of monitoring (i.e. the number of variables measured, the frequency of measurement). Model predictions of prevailing particle trajectories could be used to establish the locations of sampling sites required to identify areas of greatest predicted impact. Model predictions might also be used to establish the extent of a benthic mixing zone. Some regulatory authorities have accepted degradation within a specified area around a given farm and directed monitoring efforts at the perimeter of the mixing zone in order to verify compliance.

Estimates for the waste loading over a potential area affected, *i.e.* waste dispersed distance (D) can be derived from the following equation:

$$D = \frac{z \times v}{S_{(1 or 2)}} \qquad (2.2)$$

Where:

z is the depth of water (m),

v the current velocity (m s^{-1}),

S_1 and S_2 the settling velocities of uneaten food and faecal waste respectively (m s^{-1}) (from Gowen *et al.*, 1989; see Figure 2.3).

Solid waste dispersion based on the principles as following:

- faecal pellets and uneaten food particles fall at a rate s through a depth z take a time $t = z/s$ to reach the bottom
- during time t, current velocity v will displace the particle horizontally by a distance $D = vt = v\,z/s$ from the cage

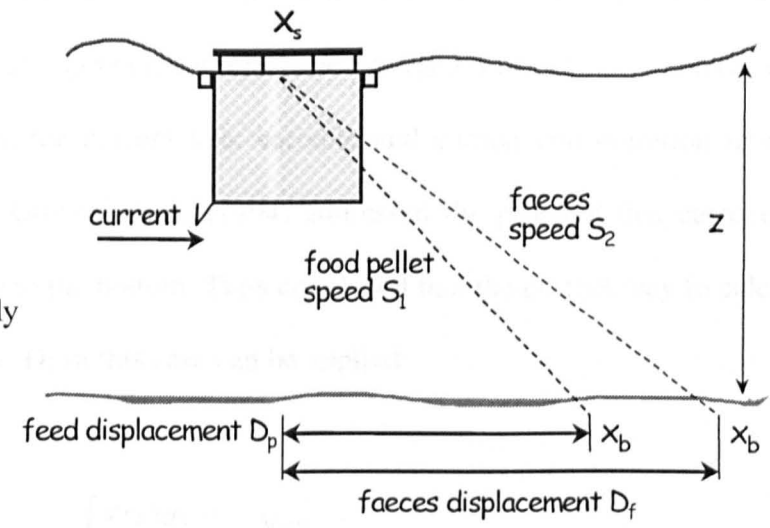


Figure 2.3 The principle and formulation of solid waste dispersion model (from Gowen *et al.*, 1989)

However, the waste dispersion model in the past ten years has relied heavily on some assumptions regarding uniform depths and single values for the sinking velocity of feed and faecal pellets (Gowen *et al.*, 1989; Silvert, 1994). Nevertheless this has led to the development of more complex and comprehensive models and increasing attention has been paid to the nature of the waste itself. Development of dispersion models has been hindered by a lack of adequate quantitative information on physical and chemical characteristics of waste particles (Silvert and Sowles, 1996). In order to deal with these difficulties a variety of approaches has been used. However, none has proved totally satisfactory, although several important patterns have emerged. For example, in most cases with respect to tidal inlets, the current v is variable and carbon concentration must be averaged over time. Thus, Gowen *et al.*, (1994) addressed the problem that currents are seldom uniform all the way to the bottom. They concluded that the correct way to calculate the horizontal displacement (D) in this case can be applied:

$$D = \frac{\int V(z)d(z)}{s} = \frac{\mathrm{Vav}}{\mathrm{s}} \qquad (2.3)$$

where

V is the uniform current speed (m s^{-1})

z is the depth (m)

V_{av} is the depth-average value of V (m s^{-1}),

s is the settling velocity of particulate waste (m s^{-1}).

In addition, settling velocities of feed and faeces may be assigned mean values (Gowen *et al.*, 1989) or should be treated as a probability distribution with a defined mean and

standard deviation (Hagino, 1977 in GESAMP, 1996). Much more refined models can be developed if sufficient biological and physical data are available. The main limitation to using this type of model is not mathematical, but rather the availability of data and the time and resources needed to do the calculations (Silvert and Sowles, 1996).

It is important to distinguish between food and faeces particles since the settling velocities of the two are different. It is likely that there are widespread size and density spectrums of waste particles and hence settling velocities (Barg, 1992). One of the most important determining factors is the relationship between the settling velocities and waste particle sizes and will thus be investigated in this thesis. More thorough consideration of various aspects such as possible consumption of uneaten food by wild fish, possible resuspension of sedimented materials, differences in bottom characteristics, effects of benthic organisms and other microbiological and chemical processes on the deposited organic particulate matter, should be accounted for in any models (see Holmer and Kristensen, 1992; Gowen *et al.*, 1994; Silvert, 1994, Sowles *et al.*, 1994). Existing waste dispersion models make no prediction on the ecological consequences of a given loading. Linkages between predicted loading and biological response (*e.g.* changes in population densities, species number, a functional response of the community) are not yet established and are likely to be highly site-specific (GESAMP, 1996). Although the initial steps of relating carbon loadings with sea-bed oxygen consumption and CO_2 production have been taken by Findlay and Watling (1994), much work remains to be done before dispersion modeling can be used to quantify the loading that would maintain a given ecological state (GESAMP, 1996).

Part of the information contained in Chapter 3 has been published in Aquaculture International – Chen, Beveridge and Telfer 1999, 7 (88--100). Edited by Poxton M. G. and published by Chapman & Hall Ltd., whereas in Section 3.2.6 and 3.3.6 has been published in Aquaculture Research – Chen, Beveridge and Telfer 1999, 30 (395-398).

Chapter 3
Physical characteristics of commercial pelleted fish feeds and faecal matters

3.1 Introduction

Regulation of cage mariculture increasingly relies upon predictive modeling of waste outputs and dispersion, and upon the application of environmental capacity criteria (Beveridge, 1996). Waste dispersion modeling generally has two components. First, mass balance principles are used to quantify wastes, after which simple settlement and dispersal algorithms are used to characterize dispersal (Gowen and Bradbury, 1987; Gowen, 1994; Elberizon and Kelly, 1998). Although nutrient dispersal models take account of both food and faeces, uneaten food is likely the greatest contributor to loadings of solid waste (Beveridge *et al.*, 1991; Gowen *et al.*, 1994; Hevia *et al.*, 1996). The proportion of food fed that is not ingested ranges from 1 to 40%, although values of 5 to 15% are most often reported (Thorpe *et al.*, 1990; Juell, 1991; Blyth *et al.*, 1993; Findlay and Watling, 1994; Wu *et al.*, 1994; Beveridge, 1996; Beveridge *et al.*, 1997; Cho and Bureau, 1997).

The physical quality of feed pellets is important for a number of reasons. First of all, transportation and handling in both factory and on farm situations require pellets of an integrity sufficient to withstand production of fines through attrition stresses (Thomas and van der Poel, 1996). Pellets of high physical quality must also not compromise nutritional quality. The physical properties must promote maximum feed intake and allowed greatest possible nutritional value to be extracted. Two components of physical quality are of particular importance: hardness and friability. Hardness is defined here as the force necessary to crush a pellet or a sample of pellets; friability is defined as the amount of fines lost from the pellets after being subjected to mechanical agitation. Hardness is a quality which is important for the nutrition of animals since it can influences food preference (Skoch *et al.*, 1983). Different animal species require

different physical properties with respect to feed pellets. Many fish species, for example yellowtail (*Seriola quinqueradiata*) do not readily accept food pellets (Viyakarn *et al.*, 1992; Shimeno *et al.*, 1993).This means that different standards are necessary. In terms of environmental impacts of waste feeds, pellet characteristics such as sinking velocity and water stability are important (Elberizon and Kelly, 1998; Chen *et al.*, 1999b).

The other main source of particulate wastes from fish farms is faecal matter. In order to assess the impacts of caged Atlantic salmon *Salmo salar* L. farming on the marine environment and match production to site capacity, it is essential to know not only the quantities and composition of uneaten feed and faecal wastes but also their physical characteristics such as settling velocities to incorporate into the waste dispersion model. However, waste dispersion models applied to the cage culture of Atlantic salmon (*Salmo salar* L.) have relied on scant data sets of feed settling values (Gowen and Bradbury, 1987) which take no account of the physical characteristics of fish feed (pellet type, pellet size), environmental conditions (temperature, salinity) and feed formulation, there are also no data for other marine fish feeds other than mainstream salmon feeds. Current range of commercial salmon diets include both 'standard' and 'high-energy' formulations (Springate, 1991; Cho *et al.*, 1994; Anderson *et al.*, 1996; Heinen *et al.*, 1996). A similar situation is apparent with the way in which faecal pellets are accounted for in waste dispersion models. It is assumed that the settling rate for faeces is constant, irrespective of the environmental conditions, feed formulation, and faecal pellet size. Thus, all faeces settle at a 'standard' rate. Despite this, there has been little research on fish faeces. In this study we determine physical characteristics and settling velocities of a range of commercial feeds including Atlantic salmon (*Salmo salar* L.), sea bass (*Dicentrarchus labrax*) and sea bream (*Sparus aurata*) feeds under defined laboratory

conditions as well as the settling velocities of salmon faecal pellets, collected by different methods, and produced by fish fed different feed formulations so that data may be incorporated into models for more accurate prediction of dispersion of solid wastes.

3.2 Materials and Methods

Atlantic salmon feeds produced by two major companies, Ewos and Trouw Aquaculture, were selected. Diets included a comprehensive range of pellet sizes and both 'standard' (20 to 24 % oil) and 'high energy' (28 to 30 % oil) (HE) diets (Table 3.1). Sea bass and sea bream feeds were supplied by ProAqua (Table 3.2). Feed samples from single batches were used throughout. Salmon faecal pellets were obtained from the University Marine Environmental Research Laboratory, Machrihanish, Argyll, UK.

3.2.1 Hardness and friability of feed pellets

The equipment used to determine pellet hardness and friability was designed and built according to feed industry specifications. The methodologies followed those described for industry by Thomas and van der Poel (1996). To test hardness, ten pellets of different sizes of salmon diets (Ewos: 6 mm, 8 mm, 10 mm, HE 6 mm, HE 10 mm; Trouw: 6 mm, HE 6 mm, HE 8.5 mm, HE 11 mm), sea bass diets (ProAqua 5 mm, 7 mm), sea bream diets (ProAqua: 4.5 mm, 5 mm), which were within the size range of the test equipment (Kahl pellet hardness tester, 4-11 mm), were placed individually in a pellet crusher and the pressure at which the pellet disintegrated recorded (Thomas and van der Poel, 1996). An attrition mill was used to determine friability (Pfost, 1963 in Thomas and van der Poel, 1996). Samples (50 g) of salmon feed from Ewos (standard

Table 3.1 Atlantic salmon pellets used (+) in settling velocity trials.

Diet / Diameter (mm)	2	4	6	8	8.5	10	11	14
Ewos								
standard[1]	+	+	+	+		+		
high-energy[2]			+			+		
Trouw								
standard[1]			+					+
high-energy[2]		+	+		+		+	

[1] standard formulations contain 20-24% total lipid, 42-46% protein.

[2] high-energy formulations contain 28-30% total lipid, 40-46% protein.

Table 3.2 Sea bass and sea bream pellets used (+) in settling velocity trials.

Diet / Diameter (mm)	1	2	3	4.5	5	7
Sea bass [1]		+	+		+	+
Sea bream	+[2]		+[2]	+[3]	+[2]	

[1] Sea bass formulations contain 21% total lipid, 47% protein.

[2] Sea bream formulations (3 mm and 5 mm) contain 17% total lipid, 44% protein.

[3] Sea bream formulations (4.5 mm) contain 12% total lipid, 45% protein (steamed diets).

diets 2 mm, 4 mm, 6 mm, 8 mm, 10 mm) and Trouw (standard diets 6 mm, 14 mm; HE diets 4 mm, 8.5 mm, 11 mm), sea bass diets (2 mm, 3 mm, 5 mm, 7 mm), and sea bream diets (1 mm, 3 mm, 4.5 mm, 5 mm) were placed in the attrition mill box and the box revolved at 50 rpm for 10 min. Samples were then sieved and the proportion of the original sample weight that passed through a laboratory sieve (mesh size 2 mm, except the 1 mm sea bream diets were sorted using a 750 μm sieve) recorded.

3.2.2 Determining water stability of fish diet pellets

To determine the weight loss of artificially commercial salmon diet pellets in water, 25 g samples of pellets (Ewos: 2 mm, 6 mm, HE 6mm; Trouw: 6 mm, 14 mm, HE 6mm; two replicates per diet) were selected that facilitated comparison of water stability between different diet types (*i.e.* HE and standard diets) and two extreme diet sizes (smallest and largest pellets; Ewos: 2 mm and Trouw: 14 mm). Samples were placed in glass beakers adding 100 ml of sea water at 15 °C and salinity level 33 psu, approximate a typical Scottish cage culture condition during summer. Pellets soaked in sea water were left to stand with occasional gentle shaking, *i.e.* for 20 sec every 2 min, for 2.5, 5, 10,15 and 20 min, after which the samples were passed through a 2 mm sieve. Each pellet type was studied in duplicate. The collected material retained on the sieve was placed on a pre-weighed foil tray and placed in the oven (Gallenkamp OVE 300) at 105 °C for 24 h to dry and then reweighed (Mettler, model AJ 100; precision 0.1 mg). The retained dry weight was expressed as a percentage of the original dry weight according to the calculation from the moisture (M %) determination as following:

$$M(\%) = \frac{a-b}{a} x100\% \qquad (4)$$

where:

a is the un-soaked pellet weight (g),

b is pellet dry weight after 24 h drying (g).

3.2.3 Pre-test of required fall distance for final sinking velocity of fish pellets

In order to determine the distance required for pellets to attain their terminal settling velocity, a 50 cm length glass cylinder (500 ml) was first selected and separated into 4 intervals of 8.1 cm. The settling rates were carried out by timing 10 pellets (Trouw 6 mm) descent between two points with stopwatch, the first of which was 5 cm below the water surface (*c.f.* Robison and Bailey, 1981). The results indicated that the settling rate was most rapid in the latter interval (Table 3.3 also see Section 3.3.3). Hence, a 2 m length of plastic tube, diameter 10 cm, was chosen as the retest of sampling distance. The transparent tube was labeled outside every 10 cm and divided into four intervals (first three intervals each 50 cm, the last of 40 cm distance). Freshwater was poured into the tube and four types of fish pellets (Table 3.4) were placed gently with forceps just below the water surface. The settling rates were carried out as described above by timing the descent of pellets between two points of each interval.

3.2.4 Settling velocity of un-soaked fish pellets

A 1.25 m length of 10 cm diameter perspex tube was finally chosen for determination of pellet sinking velocity using sea water as the test medium. The transparent tube was marked every 10 cm and either secured in a vertical position or fixed with a support stand (Figure 3.1) that could be moved as required to different environmental condition requirements (*i.e.* different temperatures).

Table 3.3. Preliminary trials of settling velocity (cm s^{-1}; Mean ± 1 S.D.) of salmon fish diet (Trouw Select 6 mm) in four intervals of the 0.5 m long cylinder column.

Diet/Interval	1st interval	2nd interval	3rd interval	4th interval
Trouw Select 6mm	7.34 ± 0.56	8.07 ± 1.22	8.11 ± 0.86	8.47 ± 0.75

Table 3.4. Preliminary trials of settling velocity (cm s^{-1}; Mean ± 1 S.D.) of four salmon fish diets in four intervals of the 2 m long water column.

Diet/Interval	1st interval	2nd interval	3rd interval	4th interval
Trouw HE 6mm	11.53 ± 1.00	12.04 ± 1.19	11.64 ± 0.79	12.25 ± 0.83
Trouw Std 6mm	13.35 ± 0.80	13.80 ± 1.02	13.65 ± 0.89	13.45 ± 1.22
Trouw HE 4mm	10.56 ± 0.77	10.68 ± 0.69	10.27 ± 0.65	10.41 ± 0.98
Ewos Std 4mm	8.48 ± 1.00	8.68 ± 0.97	8.68 ± 0.76	9.10 ± 0.93

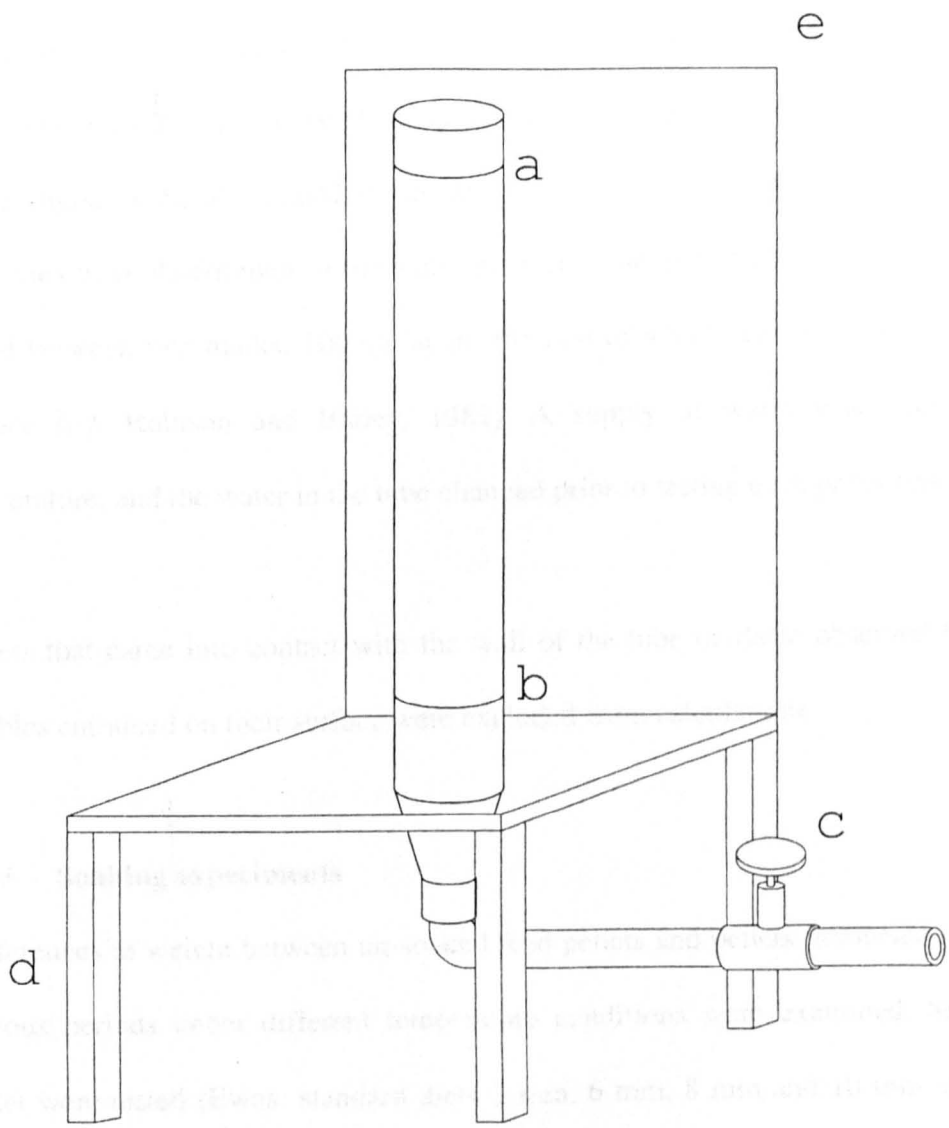


Figure 3.1 An illustration of the settling column design.

a: Start, b: Finish, c: Valve, d: Stand, e: White PVC board

Distance between the interval of a and b is 1.0 m, the internal diameter of the column is 0.1 m.

The settling rates of salmon diets listed in Table 3.1 were determined at temperatures of 10 (±1) and 20 (±1) °C and two salinities (20, 33 psu) whereas sea bass and sea bream diets (Table 3.2) were determined at 20 (±1) °C and three salinities (20, 33, 40 psu) which approximate respective *in situ* aquaculture environments. Thirty pellets of each type were taken at random, weighed, and the maximum lengths and diameters measured using digital calipers (digiMax, model m2000; precision 0.1 mm). The settling velocities were determined in the same manner as described above, the descent being timed between two marks, 100 cm apart, the first of which was 5 cm below the water surface (*c.f.* Robison and Bailey, 1981). A supply of water was stored at each temperature, and the water in the tube changed prior to testing each pellet type.

Pellets that came into contact with the wall of the tube or those observed to have air bubbles entrained on their surface were excluded from calculations.

3.2.5 Soaking experiments

Differences in weight between un-soaked feed pellets and pellets immersed in water for various periods under different temperature conditions were examined. Six types of pellet were tested (Ewos: standard diets 2 mm, 6 mm, 8 mm and 10 mm, and Trouw: HE diet 4 mm, standard diet 6 mm). Following weighing, four pellets of each type were placed in individual compartments of 24-well multi-dishes containing seawater (33 psu). Pellets were then left for periods of 0.5, 1, 3, 6, 9, 12 and 15 min at 10 (±1) and 20 (±1) °C , respectively. Pellets were then removed, placed on absorbent paper for a few moments to remove excess water, re-weighed and the increase over dry weight calculated as a percentage of the dry weight.

3.2.6 Settling velocity of immersed feed pellets

Some of the salmon pellet types (Ewos: standard diets 2 mm, 6 mm, 8 mm and 10 mm, and Trouw: HE diet 4 mm) were pre-soaked in seawater (33 psu) for 1 min to 15 min at 10 (±1) °C, and settling velocities then determined as described in Section 3.2.4.

3.2.7 Settling velocity determination of fish faecal pellets by different collection methods

Trials were conducted at the University Marine Environmental Research Laboratory, Machrihanish, Argyll, UK, during September 1997. Fulmar 50 expanded diet pellets (5.0-6.0 mm diameter, 46 % protein, 28 % oil, ME 23.6 KJ/kg), recommended for 400-1500 g salmon, were used. Ten days prior to each trial, fish were transferred from stock tanks to four 2 m diameter cylindrical tanks, within an ambient temperature 14-15 °C seawater re-circulation system, in order to acclimate them to the trial diet. Two tanks were stocked with 11 fish (mean weight = 1 kg; stocking density = 3.5 kg m^{-3}) and two with 30 fish (mean fish weight = 0.7 kg; stocking density 6 kg m^{-3}). Fish were hand-fed at a rate of 0.5% body weight day^{-1}, three times per day. It was observed that fish begin to evacuate faeces approximately 15 min after commencing feeding. Faecal pellets used in the determination of settling rates were obtained by collecting newly evacuated material with a fine mesh hand-net. Material was also collected by gently squeezing the abdomen of anaesthetised fish (100 ppm 2-phenoxyethanol solution for approximately 3 min), from the ventral fins to the anus, as described by Austreng (1978). Faecal pellets were transferred to sterile petri-dishes and stored at –20 °C. Prior to determining settling velocities, pellets were thawed, weighed, measured using digital calipers and the settling velocity determination carried out as described in Section 3.2.4. Water was maintained at 15 (±1) °C throughout, and trials conducted at two salinities (20 and 33 psu).

3.2.8 Settling velocity determination of fish faecal pellets fed with different diets in different seasons

The study consisted of three experiments conducted in December 1997, March and May 1998, at the Machrihanish Marine Environmental Research Laboratory, as above. The food pellets used for feeding salmon were Trouw expanded high energy (HE) 6 mm diets (30% oil) and Trouw standard 6 mm diets (20% oil) (see Section 3.2), supplied in 6 mm pellets suitable for feeding 400-1500 g salmon. Equipment and preparation of each trial were similar to Section 3.2.7. Fish were selected from stock held on site at Machrihanish. Prior to each trial, fish were transferred from stock tanks to four, 2 m diameter experimental tanks supplied with sea water and were allowed to acclimate to the test diet for at least 10 days. Each experimental tank held 10 fish. Two tanks of fish were fed with HE diets and two with standard diets. Environmental conditions and fish sizes are shown in Table 3.5. The fish were fed three times per day and the daily ration was determined from body weight and water temperature in accordance with feed manufacturers instructions. Hand feeding was carried out to enable close monitoring of fish condition. Faecal settling rate determinations were made using freshly evacuated faeces, collected immediately after defecation by hand net from one tank of fish fed HE diet and one tank fed the standard diet. The procedures of determination of settling velocities were the same as described in Section 3.2.4.

3.2.9 Statistical analysis

All data sets were tested for homogeneity of variances and normality. Statistical significance of differences within measured parameters were computed from one-way or

Table 3.5 Body weight (Mean ± 1 S.D.) of fish fed on high energy (HE) and standard diets (S) and the prevailing environmental conditions in different seasons.

Sampling date and feed types	Mean water temperature (°C) and salinity (psu)	Mean fish body weight (g)
December 1997		
HE	9.2 °C/33psu	742 ± 133
S		664 ± 77
March 1998		
HE	8.4 °C/34psu	1262 ± 280
S		1198 ± 165
May 1998		
HE	10.7 °C/33psu	1709 ± 259
S		1616 ± 186

two-way analysis of variance (ANOVA) using Minitab™ version 9 statistical software (Ryan and Joiner, 1994). In all cases, differences were considered as significant where $p < 0.05$.

3.3 Results

3.3.1 Pellet hardness and friability

Results for salmon pellet hardness (pressure; kg cm^{-2}) are given in Table 3.6 and those of sea bass and sea bream pellets are given in Table 3.7. One-way ANOVAs comparing hardness of different salmon diets show significant differences ($F_{8, 81} = 98.71$, $p < 0.001$) as well as in sea bass and sea bream diets ($F_{3, 36} = 8.15$, $p < 0.001$). Observation

Table 3.6 Mean (1 S.D.) hardness values, expressed in terms of kg force cm^{-2} pellet, for Atlantic salmon feed pellets

Diet/type	6 mm	8 mm	8.5 mm	10 mm	11 mm
Ewos					
standard	2.4 (0.5)	2.7 (0.4)	-	4.7 (0.5)	-
high energy	2.6 (0.3)	-	-	3.9 (0.7)	-
Trouw					
standard	6.0 (0.5)	-	-	-	-
high energy	4.0 (0.6)	-	6.4 (0.5)	-	6.3 (0.7)

Table 3.7 Mean (1 S.D.) hardness values, expressed in terms of kg force cm^{-2} pellet, for sea bass and sea bream feed pellets.

Diet/type	4.5 mm	5 mm	7 mm
Sea bass	5.1(1.7)	3.4 (0.3)	-
Sea bream	-	4.2 (0.6)	5.1 (0.5)

of pooled 95% confidence intervals shows that there were three defined groups of pellet hardness of salmon diets. The softest pellets were Ewos standard 6 mm and standard 8 mm and HE 6 mm; of moderate hardness were Ewos standard 10 mm and HE 10 mm and Trouw HE 6 mm diets, and the hardest pellets were Trouw standard 6 mm, HE 8.5 mm and HE 11 mm diets. Under test conditions the mean hardness values of 5 mm extruded sea bream diets (3.4 kg cm^{-2}) were less than those of 4.5 mm pelleted (steamed) diets (5.1 kg cm^{-2}).

Friability of salmon diets ranged from 0.2% for HE 4 mm pellets from Trouw to 5.7% for standard 14 mm pellets from Trouw, whereas in sea bass and sea bream diets friability ranged from approximately 0.2 % to 1.2 % and 0.3 % to 1.0 % respectively (Table 3.8). Although the friability of 4.5 mm pelleted (steamed) sea bream diets was only 0.97 %, it was nevertheless higher than other three extruded sea bream diets. Moreover, there was a trend towards measuring friability with decreasing pellet size (Table 3.8).

3.3.2 Water stability of fish diet pellets

Figure 3.2 shows % loss in weight of pellet samples plotted against immersion time for six different pellet types under 15 °C - 33 psu condition. The smallest pellets (Ewos 2 mm) were least stable in water, as indicated by weight loss over time, whereas the Trouw 6 mm diets has the greatest water stability after 20 min immersion in sea water among the six diets. Nevertheless, all diets showed good stability in water, with less than 4 % loss of integrity, after 20 min immersion in sea water.

Table 3.8 Friability (%, n=2) of feed pellets of five Atlantic salmon diets produced by Ewos and Trouw and four sea bass and sea bream diets produced by ProAqua.

Diets	Friability (%)
Atlantic salmon	
Ewos 2 mm	0.58
Ewos 4 mm	0.70
Ewos 6 mm	1.03
Ewos 8 mm	3.32
Ewos10 mm	2.13
Trouw 4 mm HE	0.20
Trouw 6 mm	0.23
Trouw 8.5 mm HE	0.88
Trouw 11 mm HE	1.67
Trouw 14 mm	5.69
Sea bass	
2 mm	0.90
3 mm	0.24
5 mm	0.59
7 mm	1.15
Sea bream	
1 mm	0.29
3 mm	0.84
4.5 mm	0.97
5 mm	0.52

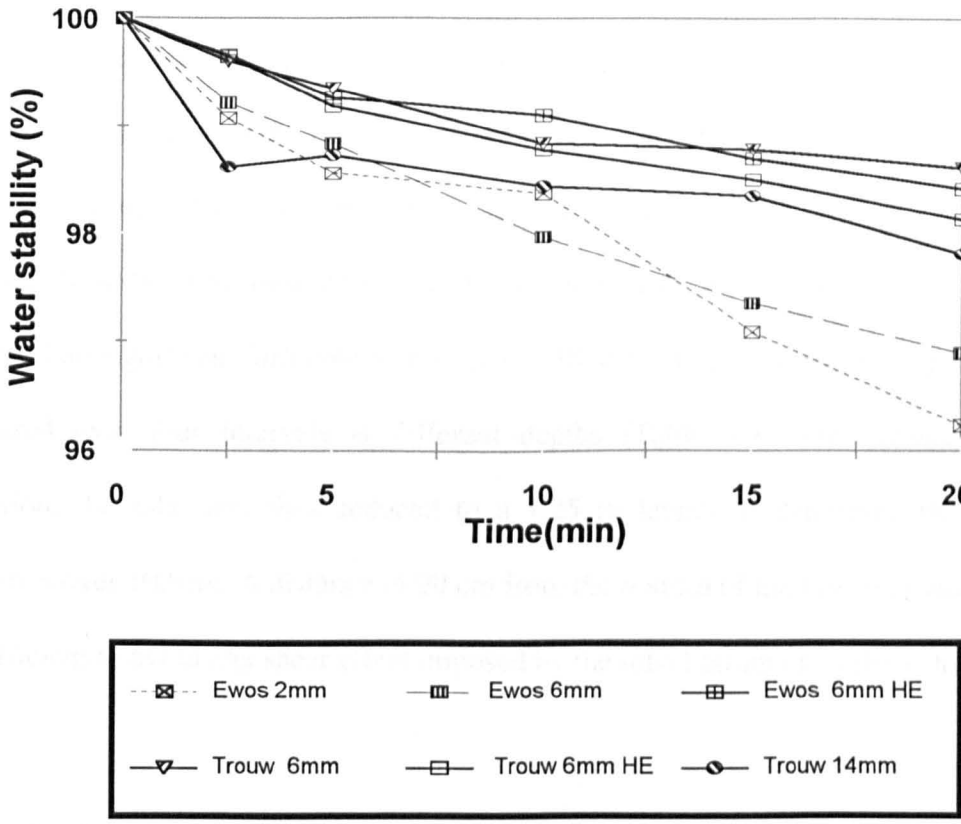


Figure 3.2 Mean water stability (%) of six salmon diets immersed up to 20 min in sea water (15 °C - 33 psu).

3.3.3 Pre-test of required fall distance for final sinking velocity of fish pellets

The results of the first preliminary trial to determine the required distance for pellets reaching terminal settling velocity using a 50 cm length cylinder are shown in Table 3.3. From the table it can be seen the settling velocities in the latter intervals were greater than those determined in the first two intervals ($F_{3, 36} = 2.93$, $p < 0.05$). It appeared that a longer settling column was needed to satisfy the distance required to reach terminal velocity. Results from trials with a range of pellet sizes in 2 m retest water column indicated no significant differences ($F_{3, 116} < 2.40$, $p > 0.05$) between settling velocities measured over four intervals at different depths (Table 3.4). For convenience of operation, the tube was then reduced to a 1.25 m length to determine the settling velocities over 100 cm. A distance of 20 cm from the bottom of the tube was assumed to be sufficient to avoid any shear effect imposed by the tube bottom on pellet velocity.

3.3.4 Settling velocities of un-soaked fish pellets

Settling velocities for the different salmon pellet types are shown in Fig. 3.3 and those for sea bass and sea bream pellet types are shown in Figure 3.4 and Figure 3.5, respectively. A two-way ANOVA indicated significant differences in settling velocity of salmon diets both between different pellet sizes and within each set of environmental conditions (temperature and salinity) for the same manufacturer (Trouw: $F_{5, 174} > 26.32$, $p < 0.05$; Ewos: $F_{6, 203} > 202.5$, $p < 0.05$). The settling velocities in water of 20 psu salinity were generally greater than those in 33 psu for any given pellet type, although differences were not always statistically significant. Generally similar results were found for the settling velocities of sea bass and sea bream diets. In water of 20 psu salinity settling velocities were significantly greater than those in 33 psu and 40 psu ($F_{2, 87} >$

121.34, $p < 0.05$), except for sea bass 2 mm pellets the differences were not statistically significant ($F_{2, 87} = 1.28$, $p > 0.05$). Under 20 °C-33psu condition, significant differences ($F_{3, 116} = 1045.3$, $p < 0.001$) in the settling velocities of sea bream pellets were found in the order 4.5 mm > 3 mm> 5 mm > 1 mm (Figure 3.4) whereas in sea bass pellets were 3 mm > 7 mm > 2 mm > 5 mm ($F_{3, 116} = 56.9$, $p < 0.001$; Figure 3.5).

For salmon diets, settling velocities at 10 °C were greater than those at 20 °C for certain pellet types (Ewos standard diets: 4 mm and 6 mm pellets; Ewos HE diets: 6 and 10 mm; Trouw HE diets: 11 mm; Trouw standard diets: 14 mm) (Figure 3.3).

For Ewos, pellet settling velocities of the standard diet range increased significantly with pellet size from 2 to 10 mm ($F_{4, 145} > 271.58$, $p < 0.001$, Figure 3.3). Furthermore, regression analysis for these pellets showed strong linear relationships between pellet size and settling velocity ($R^2 > 0.94$, Figure 3.6), independent of temperature and salinity. There were no significant differences between settling rates for Ewos 6 and 10 mm HE pellets at either salinity at 10 °C ($F_{1, 58} < 1.13$, $p > 0.05$) but there were significant differences at 20 °C ($F_{1, 58} > 5.01$, $p < 0.05$) at both salinities. For Trouw HE diets, only the settling velocity of the 4 mm pellets was significantly different from the remaining sizes for HE diets ($F_{3, 116} = 33.6$, $p < 0.001$, Figure 3.3).

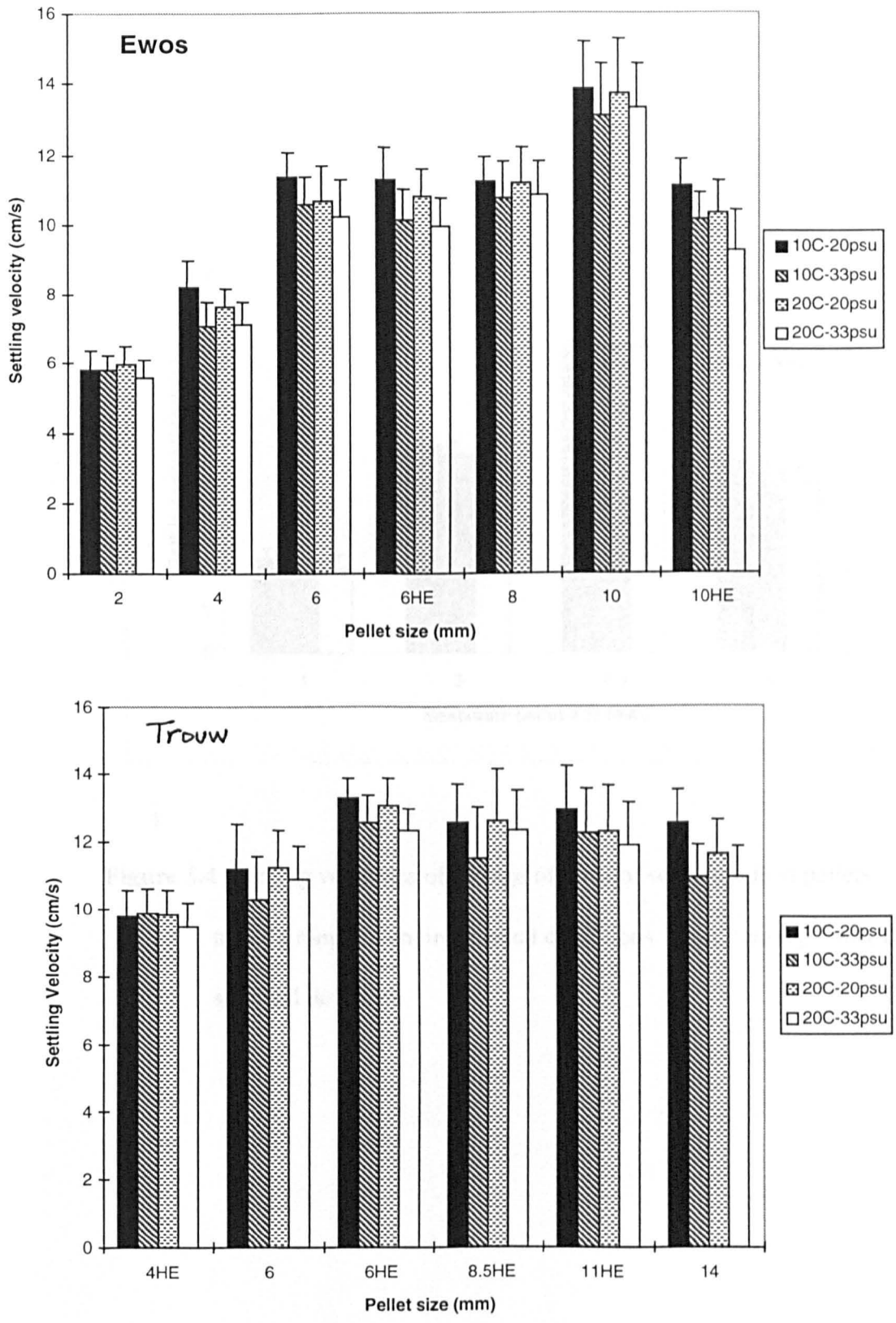


Figure 3.3 Settling velocities of Atlantic salmon feed pellets, which differ in size and diet formulation, from two manufacturers (Ewos and Trouw), under a range of environmental conditions. Error bars represent 1 standard deviation.

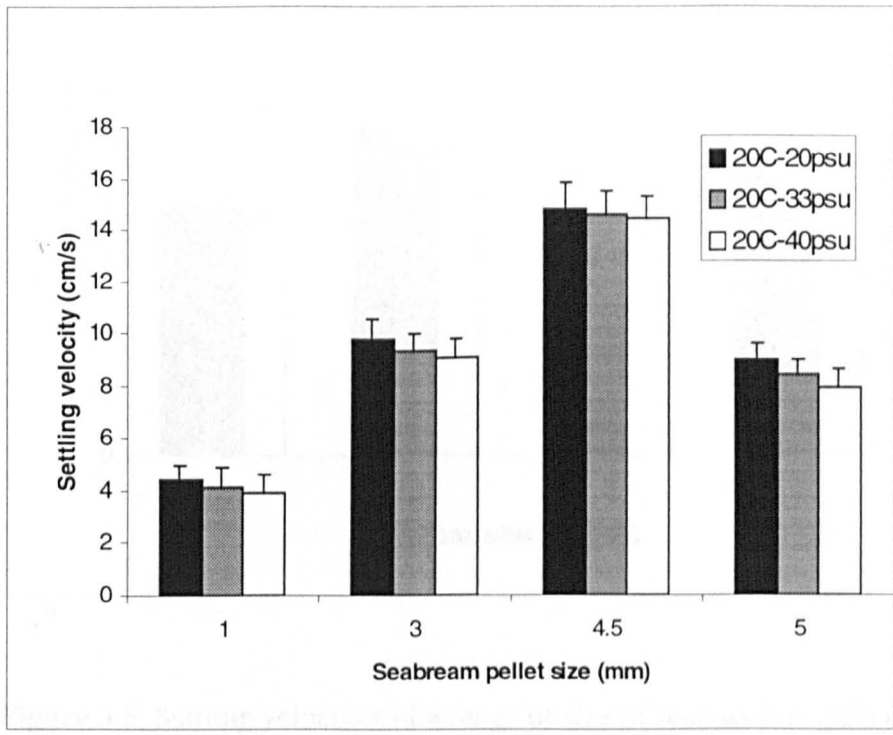


Figure 3.4 Settling velocities of a range of sizes of seabream feed pellets under a range of environmental conditions. Error bars represent 1 standard deviation.

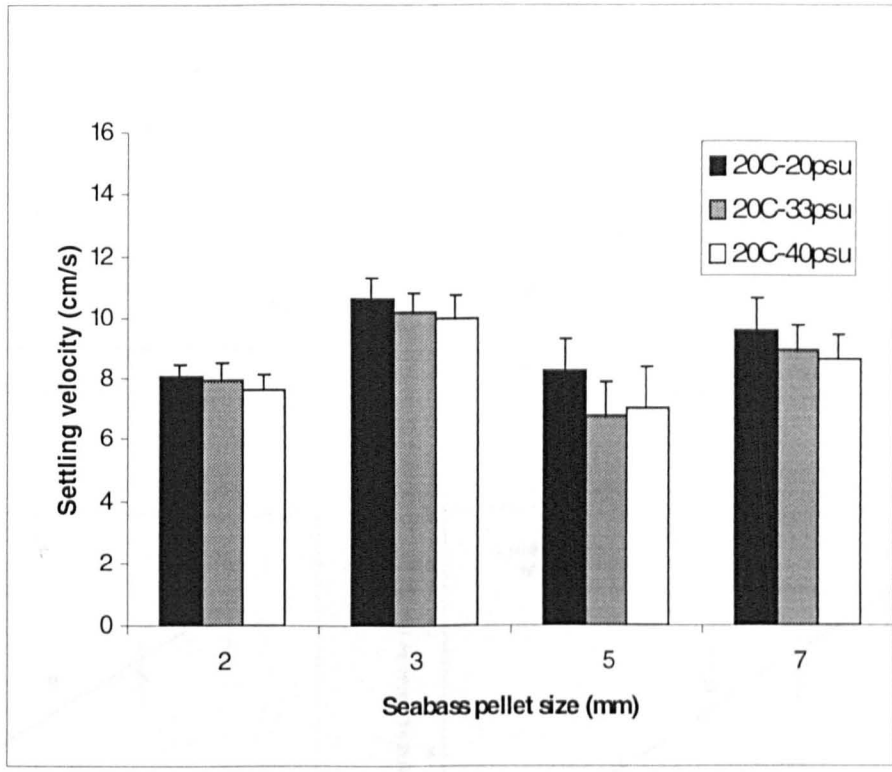


Figure 3.5 Settling velocities of a range of size of seabass feed pellets under a range of environmental conditions. Error bars represent 1 standard deviation.

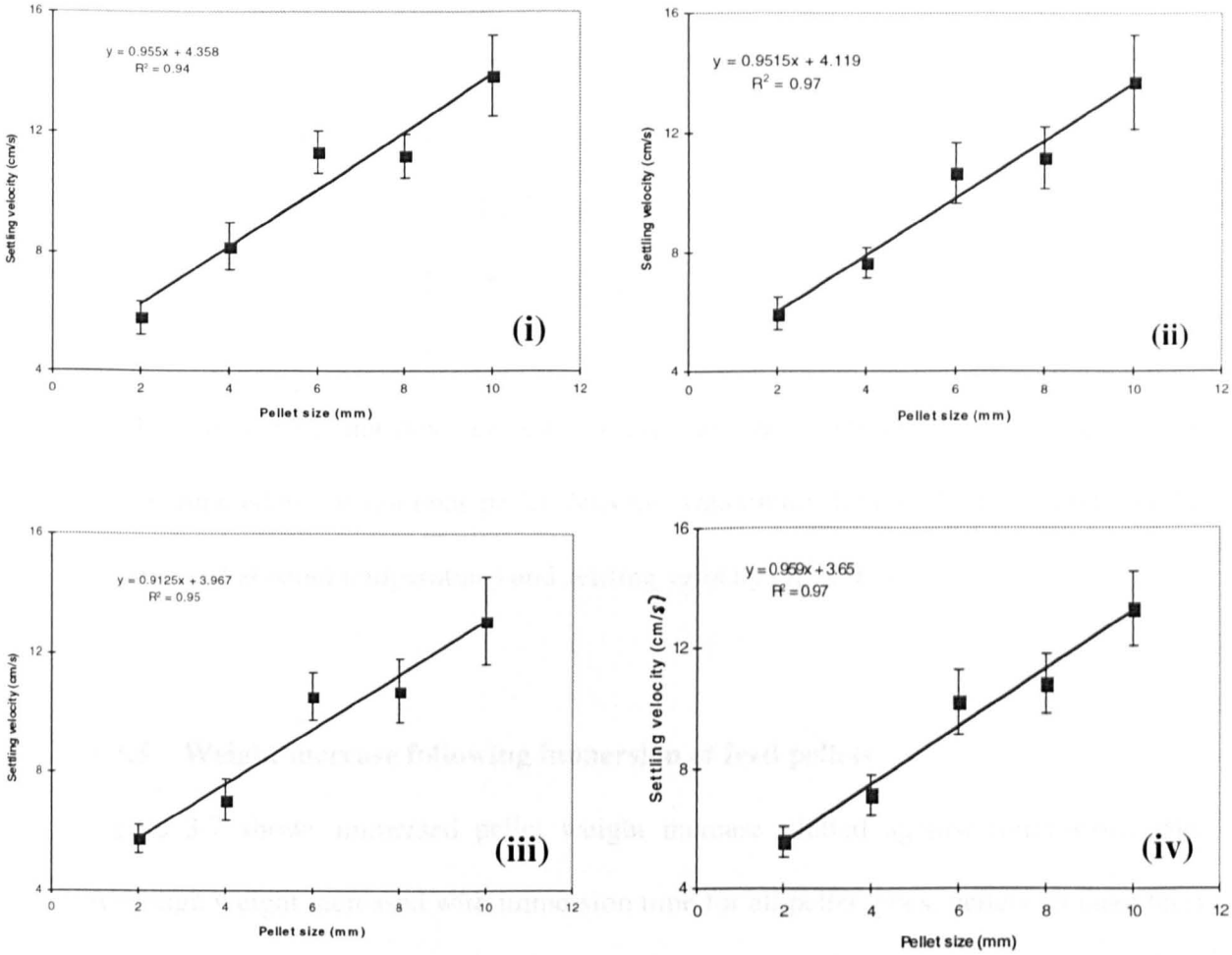


Figure 3.6 Relationships between pellet size and settling velocity under (i)10 °C - 20 psu (ii) 20 °C - 20 psu (iii) 10 °C - 33 psu (iv) 20 °C - 33 psu for Ewos standard salmon diets. Error bars represent 1 standard deviation.

Comparison of standard and HE diets showed significant differences in settling velocity between the 10 mm standard and HE diets from Ewos ($F_{1, 58} > 94.46$, $p < 0.001$; Figure 3.6) and between the 6 mm standard and HE diets from Trouw ($F_{1, 58} > 64.05$, $p < 0.001$). However, no significant differences ($F_{1, 58} < 4.02$, $p > 0.05$) were found between 6 mm standard and HE pellets from Ewos.

Within the same pellet type (*i.e.* size), there was low correlation between variation in pellet dimensions or nominal pellet densities (maximum length, diameter and weight, determined at room temperature) and settling velocity ($R^2 < 0.3$).

3.3.5 Weight increase following immersion of feed pellets

Figure 3.7 shows immersed pellet weight increase plotted against immersion time. Although weight increased with immersion time for all pellet types, pellets retained their shape. A paired t-test showed no significant difference between the increases in pellet wet weight observed at the two temperatures ($t < 0.39$, $n=14$, $p > 0.05$). The greatest and fastest weight increase was apparent for the smallest pellet (Ewos, 2 mm), from 10% after 0.5 min immersion time to nearly 30% following 15 min immersion. By comparison, the increase in wet weight with immersion time in the largest pellets (Ewos, 10 mm) only ranged from 1.4 % after 0.5 min to 6 % after 15 min immersion.

3.3.6 Settling velocities of immersed feed pellets

The results for sinking velocities of wet pellets are shown in Table 3.9. For the same diets, there was no significant difference in settling velocity ($F_{15, 48} < 1.41$, $p > 0.05$)

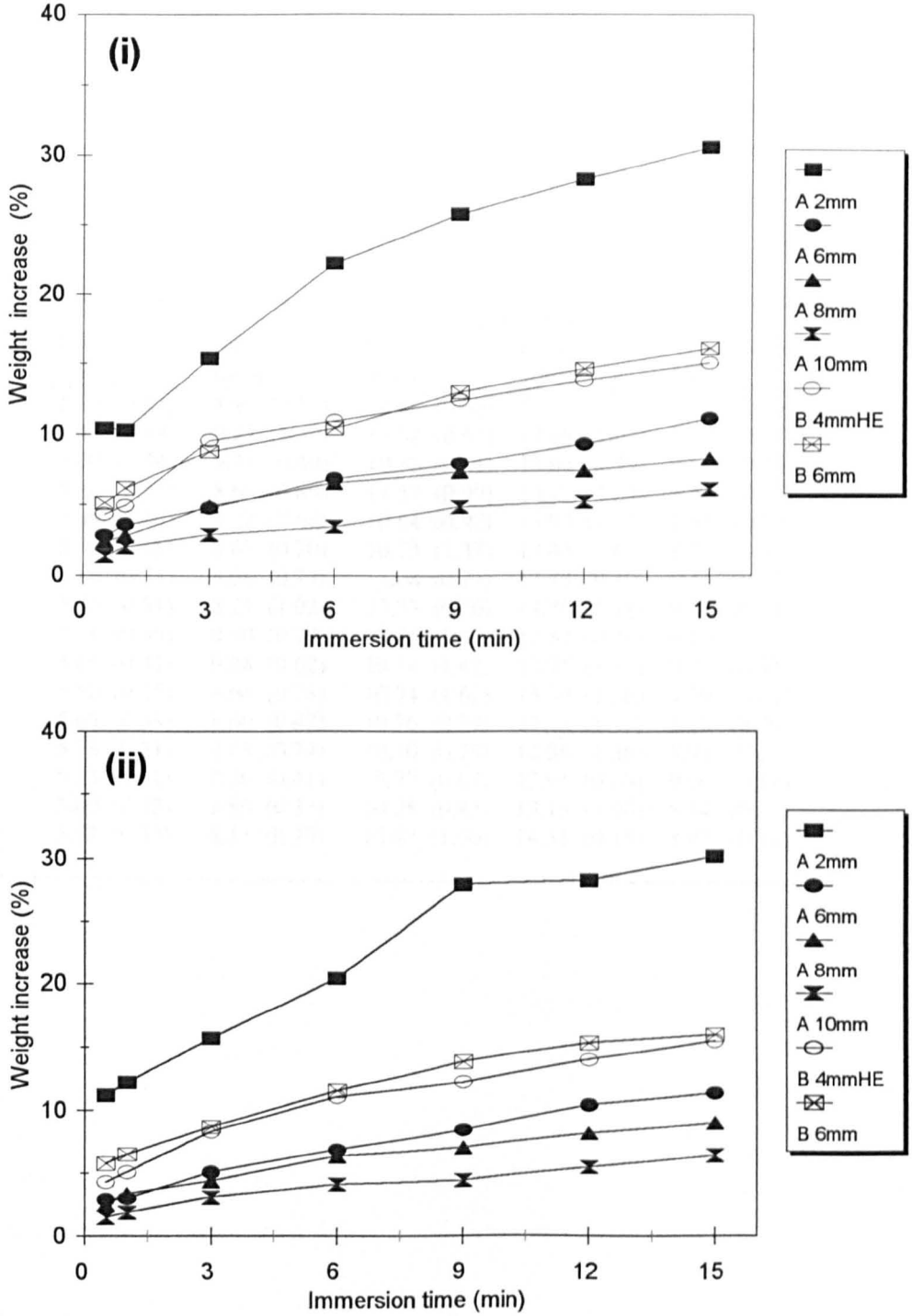


Figure 3.7 Mean weight increase (%) of pelleted Atlantic salmon feed pellets (A – Ewos, B – Trouw) as a function of immersion time at; (i)10 °C – 33 psu and, (ii) 20 °C – 33 psu.

Table 3.9 Mean (1 S.D.) settling velocities (cm s^{-1}) of feed pellets of five Atlantic salmon diets in sea water (10 °C-33 psu) under different immersion times.

	Settling velocity				
Immersion Time (min)	Ewos 2mm	Ewos 6mm	Ewos 8mm	Ewos 10mm	Trouw 4mm HE
0	5.86 (0.53)	8.94 (0.59)	10.98 (0.88)	14.91 (0.91)	9.12 (0.69)
1	5.45 (0.88)	9.22 (1.50)	11.32 (0.67)	12.69 (1.80)	8.72 (0.87)
2	6.00 (0.54)	9.27 (0.40)	10.32 (0.81)	13.67 (1.29)	8.91 (0.76)
3	5.80 (0.81)	8.68 (0.85)	11.57 (0.99)	13.13 (1.25)	9.55 (0.55)
4	5.49 (0.44)	9.22 (0.68)	10.64 (0.97)	13.93 (1.75)	8.95 (0.39)
5	5.45 (0.45)	8.63 (0.70)	10.73 (1.37)	13.43 (1.61)	8.72 (0.46)
6	5.86 (0.31)	8.58 (0.77)	11.08 (0.77)	13.73 (0.89)	9.45 (0.47)
7	5.85 (0.51)	8.27 (1.02)	10.33 (0.70)	14.70 (1.18)	9.03 (0.77)
8	5.46 (0.56)	9.50 (0.79)	10.35 (1.42)	12.37 (0.78)	9.10 (0.61)
9	5.65 (0.41)	9.28 (0.62)	10.14 (1.42)	12.91 (1.40)	9.02 (0.45)
10	5.50 (0.25)	8.64 (0.78)	10.74 (1.62)	13.39 (1.36)	9.39 (0.61)
11	5.65 (0.38)	8.60 (0.47)	10.76 (0.94)	12.73 (1.40)	8.90 (0.30)
12	5.78 (0.31)	9.48 (0.74)	10.30 (0.75)	12.56 (1.36)	8.97 (0.60)
13	5.15 (0.31)	8.30 (0.41)	9.72 (0.67)	12.94 (0.69)	9.06 (0.32)
14	5.80 (0.32)	8.89 (0.33)	10.25 (0.43)	13.15 (1.00)	8.54 (0.61)
15	5.52 (0.32)	8.11 (0.29)	10.47 (1.00)	14.31 (0.15)	8.85 (0.31)

3.3.7 Settling velocity determination of fish faecal pellets by different collection methods

Physical characteristics, including faecal pellet size and settling velocities, are shown in Table 3.10. Mean settling velocities for smaller pellets (mean length < 5 mm) collected by stripping ranged from 5.4 cm s^{-1} at 33 psu to 6.6 cm s^{-1} at 20 psu. Those for larger pellets (mean length > 5 mm) collected by net from tanks ranged from 5.3 cm s^{-1} at 33 psu to 6.3 cm s^{-1} at 20 psu. One-way ANOVAs were used to show differences between individual parameters. Results showed significant differences in settling velocity as a result of salinity ($F_{1, 46} = 12.71$, $p < 0.001$), but no significant differences in settling velocity could be attributed to faecal pellet size ($F_{1, 22} = 0.47$, $p > 0.05$) or to collection method (*i.e.* by net or by stripping) ($F_{1, 30} = 0.38$, $p > 0.05$).

Table 3.10 Physical characteristics and settling velocities (Mean ± 1 S.D.) of faecal pellets under two environmental conditions, 15 °C - 20 psu and 15 °C - 33 psu.

Collection method and Treatment	Pellet dimension[1] (mm)	Pellet weight (mg)	Settling velocity (cm s^{-1})
Net collection			
15°C - 20 psu (n= 24)	6.1 ± 2.1	87.3 ± 62.5	6.3 ± 1.2
15°C - 33 psu (n= 24)	6.8 ± 2.7	92.6 ± 56.6	5.3 ± 0.8
Stripping			
15°C - 20 psu (n=8)	4.7 ± 1.7	60.9 ± 57.1	6.6 ± 1.3
15°C - 33 psu (n=9)	4.0 ± 1.6	45.4 ± 24.4	5.4 ± 2.0

[1] Maximum length

3.3.8 Settling velocity determination of fish faecal pellets fed with different diets in different seasons

The physical characteristics and settling velocities of faecal pellets in different seasons fed with HE and standard diets are summarised in Table 3.11. Results from two-way ANOVAs performed on the data from three different monthly samples under the same experimental conditions (10 °C - 33 psu) indicated significant differences ($F_{2, 99} = 4.64$, $p < 0.05$) in settling velocity between seasons, but no significant difference ($F_{1, 99} = 0.14$, $p > 0.05$) between diet types (HE and standard diets) for each sampling time. Although the faecal pellet sizes appeared, in general, not to be associated with fish size (*c.f.* Tables 3.5 and 3.11), the pellet lengths of bigger fish were more variable than those of smaller fish.

Table 3.11 The settling velocity (Mean ± 1 S.D.) of faecal pellets from different seasons under laboratory condition 10 °C - 33 psu.

Sampling date and fish diet type	Pellet dimension [1] (mm)	Pellet weight (mg)	Settling velocity (cm s^{-1})
December 1997			
HE (n= 19)	8.4 ± 1.5	166.7 ± 55.5	5.1 ± 1.1
S (n= 10)	8.4 ± 2.3	152.0 ± 72.7	5.9 ± 1.4
March 1998			
HE (n= 21)	7.3 ± 2.8	221.1 ± 98.9	6.0 ± 1.1
S (n= 19)	7.0 ± 1.6	136.6 ± 56.9	5.4 ± 0.8
May 1998			
HE (n= 16)	7.3 ± 2.8	160.0 ± 108.1	6.4 ± 1.4
S (n= 18)	7.7 ± 1.4	196.5 ± 59.4	6.2 ± 1.1

[1] Maximum length

3.4 Discussion

Physical characteristics of a range of commercial Atlantic salmon pellets, seabass pellets and seabream pellets were assessed. The settling velocities of uneaten food are important for modelling of solid waste dispersion around cage farms. Although not all pellet types were investigated in all trials, the parameters measured allow settling velocities under a variety of environmental conditions to be incorporated into models of impacts from uneaten food.

Diets made by traditional methods of steam pelleting followed by compression, result in a dense pellet that sink rapidly in water. Modern diets made via extrusion, a process through which the feed material is moistened, pre-cooked, expanded, extruded, and dried, produces low density feed particles that sink slowly or float in water. Many finfish species such as salmonids, seabass, seabream and grouper respond quickly to a diet that is of suitable pellet size and palatability. Such pellets need to retain their physical stability in water for only a few minutes. From the present study, it is apparent that salmon diets have high water stability and low friability (*i.e.* high durability); the latter was also found in seabass and seabream diets. Smaller pellets were generally found to be more durable and less friable than larger ones, contradicting the findings of Robohm and Apelt, 1985, 1986 (in Thomas and van der Poel, 1996). In those studies, small diameter pellets (3 mm) were found to be more susceptible to breakage than larger diameter pellets (6 mm). Differences between studies can be attributed to variations in formulation and feed technology, for example 4.5 mm sea bream diets which were the only steamed diets in this investigation, the friability being only 0.97 % which was however higher than other three extruded sea bream diets. The present study also found

differences in hardness between salmon pellets produced by the two manufacturers: in general, Trouw pellets were harder than Ewos pellets. Seabass and sea bream diets were moderately hard by comparison with salmon diets produced by Trouw but generally harder than those produced by Ewos. It appeared the differences in lipid content between sea bass, sea bream and salmon diets does not affect hardness of the pellets, suggesting that the pelleting process is probably more important than feed formulation in determining hardness. Although relationships between pellet hardness and friability have been found (Wood, 1987), they are two distinct measures of physical qualities of feed pellets. Relationships between the two parameters are generally only found where the feed ingredients and pellet manufacturing processes are the same (Thomas and van der Poel, 1996).

According to Stokes' Law, the settling velocity of a particle is dependent upon its dimensions, shape, density and the viscosity of the medium. Viscosity in turn is dependent upon temperature, solute concentration and pressure. However, the pellet sizes are much larger than the limits of Stokes' Law. In order that Stokes' Law may be applied, the Reynolds Number (Re) should be less than 0.5 and hence the maximum settling velocity should be <1 cm s^{-1} (Smith, 1975). In general, sinking rates of salmon diets increased with pellet size, from a mean value of 5.6 cm s^{-1} for the smallest pellet (Ewos: 2 mm) to 13.9 cm s^{-1} for the 10 mm standard pellets produced by Ewos. The settling velocity of large, 14 mm standard diet from Trouw (10.9 cm s^{-1}) did not conform to the trend, possibly because of increased frictional drag of larger pellet size or because of differences in pellet density. Settling velocities of extruded diets for sea bream diets ranged from 3.9 to 9.8 cm s^{-1} and from 6.9 to 10.6 cm s^{-1} for sea bass diets, broadly similar to those for salmonid diets. These comparisons exclude the steamed

pellets (4.5 mm sea bream diets) that had the most rapid settling velocity (approximately 15 cm s^{-1}, Figure 3.4) observed in the present study, and are explained earlier in this section by their greater density in the producing process. There is no trend of increasing settling velocity with pellet size for sea bream and sea bass diets, which was found in Ewos salmon diets. Moreover, there were no significant differences in settling velocities between equivalent sized sea bass and sea bream diets.

Estimates, based on assumptions that pellets are cylindrical in shape, show that the surface area : volume ratio of salmon diets ranged from 1:16 for the largest pellet (Trouw: 14 mm) to 1:3 smallest pellet (Ewos: 2 mm). Weight increased more rapidly with immersion among small pellets, in line with the larger surface-volume ratio. Nevertheless, there were no significant differences in settling velocity between un-soaked and immersed pellets.

The published data on settling velocities of aquaculture feeds is scanty. Gowen and Bradbury (1987) quote results from unpublished studies of velocities of 9 to 15 cm s^{-1} (no pellet sizes given). Findlay and Watling (1994) provide data on seven North American pellet types or sizes and quote settling rates of 5.5 to 15.5 cm s^{-1} for 3 mm and 10 mm dry pellets. Elberizon and Kelly (1998) showed settling velocities of freshwater salmonid pellet diets ranged from 5 to 12 cm s^{-1} for 2 mm and 8 mm pellet sizes, respectively. The recently published study of Booth *et al.* (2000) showed settling velocity of freshwater silver perch pellet diets of 11 cm s^{-1} for 3 mm steamed pellets. These results are similar to settling rates found here under marine conditions. In the present study the use of large sample sizes (30 replicates per treatment) increases the statistical reliability of the settling velocity data (coefficients of variation ranged

between 5- 12 %, 4-19 %, 6-18 % in salmon, seabass and seabream diets, respectively). Further, the experimental water column was much deeper than that used in other particle settlement studies (Robison and Bailey, 1981; Findlay and Watling, 1994), thereby reducing influences of drag by the tube walls and resistance exerted by the tube bottom. While data across studies are similar, direct comparisons are difficult as environmental conditions under which they were collected possibly differed.

The density of sea water decreases by approximately 0.2% between 10 and 20 °C (Kalle, 1971) so it was expected that food pellets would sink more rapidly at 20 °C than at 10 °C. However, pellets sank faster at 10 °C than at 20 °C for some pellet types, perhaps because either the influence of temperature on pellet density or the methods used here unable to detect the effect of such a small change in density. Estimates were calculated of the time taken for the slowest (Ewos 2 mm) and fastest (Ewos 10 mm) settling salmon pellets to fall through a 30 m water depth, typical of most fish farm sites, at 10 °C and 33 psu, 2 mm pellets from Ewos will take approximately 8 min 39 s to fall this distance while 10 mm pellets produced by Ewos will take 3 min 50 s. For sea bass and sea bream diets under similar condition at 20 °C and 33 psu the time taken for the slowest (ProAqua 1 mm) and fastest (ProAqua 4.5 mm) will be 12 min 18 s and 3 min 25 s, respectively.

As no differences in settling rate for periods of < 15 min immersion were found, it is concluded that it is unnecessary to take into account of changes in feed pellet density upon settling velocity during sedimentation, given the depths at cage sites and the settling times involved.

It proved straightforward to collect intact freshly evacuated material either by net collecting or stripping in the present study. While it may be inappropriate to use faecal samples collected by stripping for nutrient leaching rate determinations, such samples may be used for determination of settling velocities. Fresh net-collected faecal pellets appeared to consist of fine solid material approximately the size of the formulated diets. The range of faecal settling velocities determined at 15 °C and 33 psu over a 1 m distance in a 1.25 m column was 3.7 to 6.2 cm s^{-1} (mean = 5.3 cm s^{-1}). The mean value approximates those of 4 cm s^{-1} and 6 cm s^{-1} used by Gowen and Bradbury (1987) and NCC (1990) for modelling purposes, but is at odds with the value of 2 cm s^{-1} determined over a 10 cm distance by Findlay and Watling (1994) and the range of 1.5 to 3.0 cm s^{-1} determined over a 1 m distance by Elberizon and Kelly (1998). The recently published study of Wong and Piedrahita (2000) showed settling velocity of manually stripping faecal material from rainbow trout resulted in a median settling velocity of 0.7 cm s^{-1}. These studies may be explained by differences in faecal pellet size among the studies. Settling velocities of faeces are much lower than those of feed pellets at present study (6 to 14 cm s^{-1}). For example, it would take 10 min for a 6 mm faecal pellet falling through a 30 m water column to reach the seabed, compared with 4 to 5 min for a food pellet of similar size. Faecal wastes are thus likely to be spread over a greater area of the seabed around cages than uneaten feed pellets.

Faecal pellets were observed to be of almost uniform diameter along their length. As they fall through the water they tend to disintegrate or absorb water, resulting in a decrease in settling velocity. However, neither faeces length nor diameter is a good predictor of settling velocity at present study. The differences at faecal density may play an important role in the settling rate, however, the first trial for determining faecal

settling velocities using faecal pellets collected by different methods sample fish were fed with same diet, hence the effect should be negligible. In terms of variation in formulation and seasonal effect in the second trial, there were no significant differences in the size distribution of faecal pellet lengths and pellet weights, of faeces fed either HE or standard diets. The settling velocities of faeces produced by fish fed HE diet were slightly lower than those fed by standard diet conducted in December but in contrast the settling velocities of faecal pellets fed by HE diets were slightly higher than those standard diets in March and May. Again, this may be due to the variability of pellet size. Nevertheless, no difference was found in the settling velocities of those samples. Site-specific environmental conditions (depth, currents, temperature, salinity) are more likely to be the principal determinants of the pattern of faecal deposition than fish - and thus faeces - size. Due to the limited fresh faecal samples collected from the present study, it was not possible to investigate more faecal characteristics such as specific gravity, faecal friability fed by different formulated diets. However, as the research is focusing on the waste dispersion, the nutrient content of solid wastes and its leaching which occurred immediately after evacuation will be the priority to be investigated and therefore will be studied later in Chapter 4.

Spatial distribution and sediment loading models for particulate wastes from marine fish cages have been developed by Gowen and Bradbury (1987), Silvert (1992; 1994) and Hevia *et al.* (1996). However, the models contain assumptions that require verification (Gowen *et al.*, 1994; see also Chapter 6), including the assumption that fish feed and faecal pellets settle at fixed sinking rates in all conditions. The present study provides important information for incorporation into models, which at present use a single estimated settling velocity for all feeds and faecal pellets, based on limited data.

Incorporation of results from the present study into the models, involving a range of pellet types and faecal matter, will help improve solid waste dispersal predictions.

Part of the information contained in Chapter 4 (Section 4.2.1 and 4.3.1) has been published in Aquaculture Research – Chen, Beveridge and Telfer 1999, 30 (395-398). Sections 4.2.3 and 4.3.3 have been submitted to Aquatic Living Resources – Chen, Telfer, Beveridge, and Roy.

Chapter 4

Nutrient leaching rate of fish feed pellets and faecal matters

4.1 Introduction

Today, diets for farmed Atlantic salmon (*Salmo salar* L.) are manufactured in dedicated production plants by companies that specialise in this area of feed compounding. The diets are formulated from highly digestible and nutritious components and have a balanced protein:energy ratio. The principal objective is to maximize feed utilization for growth, but increasingly manufacturers are concerned with reduction of faecal and metabolic losses thereby decreasing waste outputs (Cho and Bureau, 1997). While so called 'low pollution' diets have found favour with some statutory pollution regulatory bodies and fish farmers, the bioavailability of nutrients from uneaten food, faecal and urinary wastes from these diets has yet to be investigated (Beveridge, 1996).

Aquaculture wastes comprise both particulate material, mainly uneaten food and fish faeces, and soluble material consisting principally of carbon, nitrogen and phosphorus compounds (Beveridge *et al.*, 1991). To determine the impact of fish faecal wastes on sediments and to enable accurate modeling of faecal waste dispersion, it is necessary to quantify characteristics such as density, settling velocity, water stability and leaching rates of carbon (C) and nitrogen (N)-based key nutrients to the adjacent water. Few researchers have directly quantified faeces production due to the difficulties associated with collection and with leaching of soluble compounds during sampling (Lied *et al.*, 1982; Vens-Cappell, 1985; Findlay and Watling, 1994). Therefore, after determining the physical characteristics of feed and faecal pellets in the previous study (Chapter 3), the consequent priorities are to

estimate waste quantities, to determine their dispersal and estimate the area impacted. For an accurate estimation of waste quantities, the nutrient leaching from feed pellets and faecal pellets during the sedimentation process must be determined. While it is relatively straightforward to determine the nutrient leaching from feed pellets it is technically difficult to collect faeces without any soluble compounds leaching during the collection process (see above). Various methods of faeces collection have been employed, mostly in digestibility determination studies, including dissection, stripping, anal suction and use of metabolic chamber (see Table 4.1 for references). Various authors have also collected faeces from tanks by netting (Windell *et al.*, 1978; Spyridakis *et al.*, 1989), immediate pipetting (Spyridakis *et al.*, 1989), continuous automatic collector with rotating screens (Choubert *et al.*, 1979, 1982; Vens-Cappell, 1985; Storebakken *et al.*, 1998, 1999) and collection from the effluent outlet of the culture system (Cho *et al.*, 1982). Some methods, such as stripping and anal suction, require frequent handling of the fish causing stress and the samples collected may not be representative of naturally defecated materials. Passing effluent water through a purpose-built sample tube or drainage pipe is relatively straightforward but usually the faeces remain in water for 8-16 h before collection and thus some leaching of soluble material is likely to have occurred.

The present study first establishes the faeces collection technique by dissection method to minimize the error of leaching during the sampling process and ensure collection of sufficient representative faecal samples for leaching determination. A preliminary study sets out to determine the region of the hind-gut that contains material representative of freely-

Table 4.1 Summary of main methods of fish faeces collection

Collection Method	Reference
Dissection	1, 3, 5, 8, 12, 13
Stripping	1, 2, 3, 8, 9, 10, 11, 13
Suction	4, 8, 13
Metabolic chamber	6, 7

1. Allan *et al.*, 1999. 2. Anderson *et al.*, 1995. 3. Austreng, 1978. 4. Brown *et al.*, 1985. 5. Henken *et al.*, 1985. 6. Smith, 1971. 7. Smith *et al.*, 1980. 8. Spyridakis *et al.*, 1989. 9. Storebakken *et al.*, 1998. 10. Vens-Cappell, 1985. 11. Weatherup and McCracken, 1998. 12. Wilson and Poe,1985. 13. Windell *et al.*, 1978.

voided faeces and that may be used in nutrient leaching rate studies. The subsequent study undertaken gathered data on feed and faecal characteristics and determined leaching rates for C and N from feed and faecal matter produced by Atlantic salmon fed a range of commercially available diets. The effects of seasonal variability in faecal nutrient leaching produced by Atlantic salmon were also investigated.

4.2 Materials and Methods

4.2.1 Nutrient leaching rate of fish diet pellets

Leaching of C and N compounds from fish diet pellets were simulated in the laboratory using 500 ml beakers and an automatic shaker (IKA Labortechnik HS250). Pellet samples (each ~3g wet wt) from six commercially available salmon diets from Ewos (2 mm, 6 mm, 6 mm HE) and Trouw (6 mm, 14 mm, 6 mm HE) were randomly assigned to six leaching periods: 0, 2.5, 5, 10, 15, 20 min. Samples were suspended in 200 ml of artificial sea water (33 psu), prepared from a commercial sea salt concentrate (Coralife), at 10 (± 1) °C for the different immersion periods as above, or left in the open air as a control. During immersion, beakers were kept moving at a constant velocity of 12 cm s^{-1}, approximating the mean settling velocity of the feed pellets as determined in Section 3.3.4.

After immersion, samples were dried at 105 °C for 24 h in a drying oven (Gallenkamp OVE 300). The dried material was carefully ground by pestle and mortar, homogenized, and three

replicate sub-samples from each sample were then taken for determination of total C and total N, as a percentage of total dry-weight, using a Perkin Elmer 2400 Elemental Analyser.

4.2.2 Preliminary composition determination of anterior and posterior hindgut faecal matters

The environmental conditions used and the feeding trials conducted were as given in Section 3.2.7 at the University Marine Environmental Research Laboratory, Machrihanish, Argyll, UK. The faecal pellets used in the determination of settling rates, in Section 3.2.7, were obtained by collecting newly evacuated material with a fine mesh hand-net. However, this was not applicable to determining nutrient content as material was subject to nutrient leaching during the collection process. Therefore, two fish were killed by transferring to tanks containing a solution of 100 ppm 2-phenoxyethanol solution for approximately 10 min. The rectal section of the gut of each fish was dissected open and the contents of the terminal 8 cm carefully removed. The faecal material was divided into eight 8-10 mm lengths, four from the proximal section and four from the distal section (approximating the sizes of the faecal 'pellets' collected by net in Section 3.2.7), dried and prepared for the determination of total carbon and total nitrogen in the same manner as feed pellets described in Section 4.2.1.

4.2.3 Nutrient leaching rate determination from faecal pellets of fish fed different diets

The study consisted of three experiments conducted in December 1997, March and May 1998, at the Marine Environmental Research Laboratory as described in Section 4.2.2. The feeding protocol and environmental conditions were the same as described in Section 3.2.8.

Fish fed the HE diet and fish fed the standard diet, were used to collect faeces by dissection. It was observed that feeding stimulated defecation, therefore, samples were collected before the first feed of the day (0930 hours). Fish were captured and killed by an overdose of anaesthetic (100 ppm 2-phenoxyethanol for approximately 10 min). Where enough material was available, the contents of the distal 4 cm of the gut was dissected out and divided into 4 pieces of approximately equal length.

Leaching trials for faecal matter were conducted using the same experimental set up as used with the fish diet pellets in Section 4.2.1. Faecal samples (0.2-0.3 g wet wt) from the four segments of hindgut from each individual fish were randomly assigned to one of four leaching periods: 0, 2.5, 5 and 10 min. The leaching periods were selected to extend over the time period for faecal settlement through a 30 m water column, which is typical for a Scottish fish farm site, assuming a mean faecal settling velocity of 5 cm s^{-1} (Section 3.3.7). Samples were suspended in 200 ml of artificial sea water (33 psu), as described earlier for 2.5, 5, and 10 min or left in the open air as a control. During immersion, beakers were shaken at a constant velocity of 5 cm s^{-1}, which is similar to the mean settling velocity of the faecal pellets. After leaching, faecal samples were oven-dried and prepared for the determination of total C and total N in the same manner as for feed pellets described in Section 4.2.1.

4.2.4 Statistical analysis

All percentage or proportional data were normalized by arcsine transformation prior to statistical analysis (Zar, 1984). Food nutrient content data were analyzed by one-way

analysis of variance (ANOVA) using immersion time as factor (0, 2.5, 5, 10, 15, 20 min). Faecal nutrient content data were subjected to one way or two-way ANOVA using diet (HE and standard) and sampling occasion (December, March, May) as factors. Nutrient leaching data were analyzed using a general linear model (GLM) approach to investigate the variation between all experimental factors, *i.e.* the leaching rate was associated with sampling occasion (December, March, May), diet (HE and standard) and immersion time (0, 2.5, 5, 10 min). The nutrient content of un-immersed fish faecal pellets was only associated with experiment number and diet to assess the effects of seasonal variability and pellet type upon the nutrient content.

4.3 Results

4.3.1 Leaching rate of total carbon and nitrogen from fish diet pellets

The amount of nutrient leached from salmon diet pellets can be expressed in terms of the differences in the total content of the pellets before and after immersion in sea water. From analysis of pellet N and C over a time span up to 20 min, there were no significant differences ($F_{5, 12} < 3.11$, $p > 0.05$) in nutrient leaching of both C and N from all six salmon diets (Tables 4.2 and 4.3). However, there was a trend for N content in smallest pellets, Ewos 2 mm, to decline with immersion time (Figure 4.2). Although no statistically significant differences ($F_{5, 12} = 1.81$, $p > 0.05$) were found, the N leaching rate for Ewos 2mm was calculated as increasing from 2.6 to 12.6 % after immersion in sea water for 10 to 20 min.

Table 4.2 Mean concentration of feed carbon content (mg g^{-1} dry weight; mean ± 1 S.D.) of six Atlantic salmon diets after immersion in sea water (10 °C-33psu) for up to 20 min.

Diets	Feed carbon concentration after immersion					
	0 min	**2.5 min**	**5 min**	**10 min**	**15 min**	**20 min**
Ewos						
2 mm	488.4 ± 2.4	483.9 ± 2.2	484.8 ± 4.3	481.2 ± 10.7	482.5 ± 12.8	481.2 ± 9.6
6 mm	527.3 ± 2.8	525.9 ± 3.8	524.3 ± 1.5	528.2 ± 2.1	523.4 ± 1.2	522.2 ± 2.6
6 mm HE	520.2 ± 2.7	524.1 ± 3.5	524.9 ± 1.5	523.3 ± 2.9	525.1 ± 2.3	525.3 ± 5.7
Trouw						
6 mm	511.9 ± 3.5	514.7 ± 6.5	514.4 ± 2.8	505.3 ± 6.5	509.2 ± 9.7	510.5 ± 1.9
6 mm HE	518.6 ± 4.0	524.2 ± 2.8	515.4 ± 12.2	523.4 ± 1.0	518.0 ± 4.1	517.2 ± 15.5
14 mm	497.5 ± 5.4	494.8 ± 12.1	493.6 ± 3.7	494.9 ± 8.4	495.8 ± 5.7	491.3 ± 12.4

Table 4.3 Mean concentration of feed nitrogen content (mg g^{-1} dry weight; mean ± 1 S.D.) of six Atlantic salmon diets after immersion in sea water (10 °C-33psu) for up to 20 min.

Diets	Feed nitrogen concentration after immersion					
	0 min	**2.5 min**	**5 min**	**10 min**	**15 min**	**20 min**
Ewos						
2 mm	83.8 ± 4.2	82.4 ± 5.3	81.0 ± 4.2	76.3 ± 2.2	80.7 ± 1.9	77.6 ± 3.1
6 mm	73.8 ± 2.2	71.4 ± 3.1	71.7 ± 1.9	73.6 ± 2.2	72.8 ± 2.4	72.2 ± 2.8
6 mm HE	70.5 ± 1.8	75.2 ± 1.5	74.1 ± 1.1	71.3 ± 1.7	69.2 ± 2.7	72.0 ± 3.5
Trouw						
6 mm	67.6 ± 1.0	67.5 ± 1.4	67.1 ± 3.4	65.8 ± 1.2	68.3 ± 1.4	68.6 ± 1.8
6 mm HE	77.2 ± 0.4	76.0 ± 0.8	75.2 ± 3.2	76.4 ± 1.7	75.3 ± 4.7	76.2 ± 1.1
14 mm	62.6 ± 4.5	65.3 ± 3.8	60.6 ± 1.8	63.3 ± 2.5	62.1 ± 2.1	59.8 ± 2.8

4.3.2. Nutrient composition of anterior and posterior hindgut faecal matters

The C and N contents of faeces, collected from different zones of the rectum by dissection are shown in Table 4.4. One-way ANOVAs performed on the data showed no significant differences ($F_{3, 8} < 2.43$, $p > 0.05$) in either C or N content within the proximal or distal fractions. Further comparison of pooled data from the distal area and from the proximal area showed significant differences in N content from both fish ($F_{1, 22} = 8.48$, $p < 0.05$; $F_{1, 17} = 14.75$, $p < 0.05$; Fish 1 and Fish 2, respectively). By contrast, no significant differences ($F_{1, 22} = 0.08$, $p > 0.05$; $F_{1, 17} = 0.05$, $p > 0.05$; Fish 1 and Fish 2, respectively) in C content were observed.

4.3.3 Nutrient content and leaching rate of fish faecal matters

Nutrient content of un-immersed fish faecal matters

The C and N content of faecal samples subjected to different periods of leaching (Tables 4.5 and 4.6) were aggregated from individual fish into seasonal means. Mean faecal C content of controls ranged from 269 to 318 mg g^{-1} dry weight among experiment groups. There were no significant differences in faecal C content between sampling occasions ($F_{2, 45} = 0.5$, $p > 0.05$), but significant differences ($F_{1, 45} = 56.23$, $p < 0.001$) were found for diet type (Table 4.5).

Mean faecal N content in the control groups ranged from 28 to 37 mg g^{-1} dry weight, and significant differences ($F_{2, 45} = 49.2$, $p < 0.05$) were detected in faecal N content among the three sampling occasions, *i.e.* N content in faeces in the experiment conducted in December was lower than in samples from March and May (Table 4.6). Moreover, N content in faeces from fish fed the standard diets were significantly greater than those in the HE diets ($F_{1, 47} = 8.49$, $p < 0.05$).

Leaching rate of fish faecal matters fed on two different diets

Nutrient leaching rates from salmon faecal pellets were calculated from the differences in nutrient content of pellets before and after immersion in sea water (Tables 4.5 and 4.6). For individual fish, although most trials showed a consistent trend of greater leaching rate associated with increasing immersion time, after pooling data from the same sampling occasion and the same diet, there were no significant differences ($F_{2, 12} = 0.52$, $p > 0.05$; $F_{2, 12} = 1.52$, $p > 0.05$; C and N, respectively) in nutrient leaching rate from 2.5 to 10 min immersion. The un-immersed control group of individual fish faecal pellets had the highest C and N content. The results from GLMs showed significant differences ($F_{3, 45} > 14.72$, $p < 0.05$) in faecal C and N by comparing control with different immersion groups, showing significant nutrient leaching occurs when faeces were immersed in sea water. In general, the leaching rate of faecal C and faecal N after 2.5 min immersion in sea water were 4 to 14 % and 9 to 16 %, respectively but reached 22 % and 26 %, respectively, after 5 min immersion in sea water in the December samples (*c.f.* Tables 4.5 and 4.6).

Table 4.4 Carbon and nitrogen content (mg g^{-1} dry weight; mean ± 1 S.D.) of faeces from different zones of the hindgut. Numerals I-IV refer to the distal half of the rectum (posterior), V-VIII: proximal half of the rectum (anterior).

Rectum segments	Fish 1		Fish 2	
	Total-C	Total-N	Total-C	Total-N
Distal				
I	316.2 ± 1.6	30.6 ± 1.8	362.0 ± 5.5	33.0 ± 0.3
II	312.5 ± 2.2	27.6 ± 0.7	346.9 ± 4.1	31.8 ± 1.2
III	319.2 ± 4.5	28.4 ± 0.6	345.6	33.3
IV	317.3 ± 2.7	31.2 ± 2.4	344.4 ± 3.5	32.2 ± 0.3
Overall mean	316.3 ± 4.4	30.2 ± 2.7	351.2 ± 8.8	32.4 ± 0.9
Proximal				
V	353.8 ± 87.5	35.0 ± 7.0	351.2 ± 0.4	34.2 ± 0.4
VI	310.3 ± 3.9	32.4 ± 0.4	353.3 ± 7.0	34.2 ± 0.1
VII	315.0 ± 3.2	35.5 ± 1.4	348.0 ± 16.6	33.4 ± 0.2
VIII	305.0 ± 1.8	33.8 ± 0.9	357.5 ± 15.1	33.8 ± 0.3
Overall mean	321.0 ± 47.8	34.2 ± 3.8	352.4 ± 11.2	33.9 ± 0.4

Table 4.5 Mean concentration of faecal C content (mg g^{-1} dry weight; mean ± 1 S.D.) from Atlantic salmon fed on high energy (HE) and standard diets (S) after immersion in sea water for 2.5 min, 5 min, 10 min during different sampling occasions.

Diets and Sampling Occasions	Faecal carbon concentration and leaching rate after immersion			
	0 min	2.5 min	5 min	10 min
S				
December 1997	317.7 ± 7.8	274.7 ± 17.5	246.6 ± 40.3	272.8 ± 13.8
March 1998	301.6 ± 13.5	289.0 ± 3.8	278.1 ± 13.9	269.2 ± 18.4
May 1998	304.7 ± 9.7	279.7 ± 11.8	280.9 ± 6.8	287.7 ± 8.5
HE				
December 1997	269.3 ± 5.0	257.4 ± 20.7	259.5 ± 5.2	243.4 ± 13.7
March 1998	274.7 ± 17.3	253.1 ± 13.3	249.2 ± 12.6	262.2 ± 19.0
May 1998	271.8 ± 10.8	246.6 ± 16.9	236.8 ± 23.9	246.0 ± 5.7

Table 4.6 Mean concentration of faecal nitrogen content (mg g^{-1} dry weight; mean ± 1 S.D.) from Atlantic salmon fed on high energy (HE) and standard diets (S) after immersion in sea water for 2.5 min, 5 min, 10 min during different sampling occasions.

Diets and Sampling Occasions	Faecal nitrogen concentration and leaching rate after immersion			
	0 min	2.5 min	5 min	10 min
S				
December 1997	29.6 ± 3.1*	24.6 ± 0.7	21.5 ± 3.6	24.1 ± 1.1
March 1998	36.4 ± 1.6	33.0 ± 1.9	32.3 ± 0.9	30.8 ± 1.8
May 1998	37.1 ± 1.9	32.8 ± 0.1	33.1 ± 1.1	33.6 ± 0.9
HE				
December 1997	28.0 ± 2.4*	24.7 ± 3.8	24.2 ± 2.7	22.3 ± 3.8
March 1998	34.5 ± 3.0	30.6 ± 2.3	29.9 ± 1.3	30.7 ± 2.8
May 1998	33.3 ± 1.1	29.8 ± 2.3	28.2 ± 2.4	29.1 ± 3.5

* Nitrogen content of control group was significantly different from two other sampling occasions.

4.4 Discussion

Given the differences in settling times between feed pellet sizes shown by various authors, it might be expected that nutrient release would be greater from smaller than from larger feed pellets. The variation in nutrient release might also be expected to be compounded by differences in surface area : volume ratios, with smaller pellets leaching more nutrients in a given time than larger ones. In the present study, the results of the feed leaching trial do not confirm this as there were no significant differences in nutrient leaching of C and N from all six salmon diets after 20 min immersion in sea water. However, this was in line with the high water stability as described in Section 3.3.2. Although there was a trend for N content in the smallest pellets (Ewos 2 mm) to decline over the immersion time (see Figure 4.2), no statistical differences were found. Hence, allowing for the time taken for the feed pellet to reach the seabed, the leaching rate of feed pellets can be considered negligible during sedimentation. Nevertheless the leaching rate of N is higher than that of C, due to its higher solubility.

It was expected that the faecal matter in the distal zone of the large intestine would most closely approximate naturally excreted faeces in terms of nutrient content. While there were differences in N content, no significant differences in C content among pellets from different areas of the rectum were found. Data must be interpreted with caution, however, as only small quantities of faeces were taken on any sampling occasion. Moreover, samples were not taken from a physiologically defined section of the intestine and the inevitably subjective nature of sampling could lead to inconsistencies in faecal sample data compiled from individually sampling fish (Brown *et al.*, 1989; Spyridakis *et al.*, 1989).

Kristiansen and Hessen (1992) estimate that 33% of ingested N is discharged in Atlantic salmon smolt faeces. However, assuming around 30% of food ingested by fish is voided as faeces (Beveridge *et al.*, 1991), the present data suggest that faecal nitrogen losses are only in the order of 12% of ingested N. Kristiansen and Hessen (1992) derived a faecal N content value of 2.3% for Atlantic salmon smolts, somewhat lower than the range determined in the present study (2.8 to 3.8%). Differences may be explained on the basis of differences between dietary nutrient content and in the collecting methods employed and also the fish size. Austreng (1978) and Windell *et al.* (1978) have claimed that because the nutrient content of faeces collected by stripping or dissection is higher than that in excreted faeces, nutrient leaching estimates should be based on egested faecal samples. However, in contrast to Austreng (1978) and Windell *et al.* (1978) we believe that it is preferable to use faeces collected by dissection of the terminal 4 cm or so of the rectum (distal hind-gut) rather than naturally egested samples for nutrient leaching determination purposes due to its rapid nutrient leaching during the collection process.

The C and N content of faeces from fish fed standard diets was generally greater than from those fed HE diets. This may be due to the lower protein (nitrogen) content and higher lipid levels in HE diets, resulting in lower faecal N content. However, measurement showed little difference in N content between HE and standard diets. The higher faecal N and C levels may be due to the higher digestibility associated with HE diets. Jayaram and Beamish (1992) showed that diets high in lipid had increased dry matter digestibility, resulting in reduced faecal N and energy losses. Hence, HE diets may indeed operate to reduce environmental impact. In the present study, the use of HE diets resulted in a 12% reduction

in faecal C content and a 8% reduction in faecal N when compared to standard diets (*c.f.* Tables 4.5 and 4.6). Windell *et al* (1978) stated that absorption of nutrients was completed before the materials moved into the distal 2.5 cm of intestine. In the present study also showed that there were no significant differences in C and N content of material collected from any part of the terminal 4 cm or so of the distal hindgut.

In the present study a rapid loss of faecal nutrients occurred 2.5 to 10 min after immersion in sea water. Total C and total N were found to leach by as much as 22% and 26%, respectively, after 5 min immersion in one sampling occasion. Although leaching of nutrients may continue for up to 4 h after defecation (Windell *et al.*, 1978), considering the time taken by faecal pellets to reach the sea bottom, leaching data collected over 10 min should be sufficient for waste modelling purposes.

Faecal pellets of salmon were found to be rich in organic material, with C:N ratios from 10 to 12 in the present study, almost twice as high as those in feed pellets from 5 to 6 (*c.f.* Tables 4.2 and 4.3). The differences in ratio may be useful in determining the origin of solid wastes.

In conclusion, the experiments demonstrate that nutrient leaching from faeces is rapid, posing problems for waste management strategies that focus on the recovery of faecal wastes to reduce environmental impact. Dispersion models used to estimate the mass and spread of nutrient waste materials make several assumptions about the characteristics of uneaten food and faecal material (Telfer, 1995). One assumption is that the nutrient content of faeces does not change as they settle. The present study shows that immersion in sea

water for 2.5-10 min results in considerable C and N losses in faecal nutrients. Models for dispersion of solid material that take no account of losses by leaching will overestimate nutrient inputs to sediments. In addition, there may be an important input of dissolved nutrients to the water column.

Part of the information contained in Chapter 5 (Section 5.2.2 and 5.3.4) has been accepted for publication in Asian Fisheries Science– Chen, Beveridge and Telfer,

Chapter 5

Sedimentation characteristics and feed pellet resuspension

5.1 Introduction

Solid wastes resuspension

The various models which have been developed to predict the dispersion and loading of particulate waste from fish farms all employ the same conceptual approach, i.e., they use models to predict the areal dispersion and loading of organic waste from floating cage farms based on the principle that the dispersion of particles is a function of current flow and water depth (Gowen *et al.*, 1994). The models also contain a number of assumptions which may not be hold true, and hence there are varying degrees of error associated with each prediction. Most of the factors involved have been taken into account in some models, including post-depositional changes (Fox, 1990), variation in bottom topography (Fox, 1990; Hevia *et al.*, 1996), changes in current speed and direction with time and with depth (Gowen and Bradbury, 1987; Fox, 1990), quantities of waste (Gowen and Bradbury, 1987; Silvert, 1992) and settling velocities of waste particles (Gowen and Bradbury, 1987; Fox, 1990; Silvert, 1994). Regarding post-depositional changes, the Fox model (1990) only deals with changes in the form of decomposition of organic carbon. However, few of the models incorporate resuspension and transport of material (*e.g.* Cromey *et al.*, 1997, 2000), and this is recognized as possibly invalidating many of the predictions (Gowen *et al.*, 1994). As settled pellets under the cages may be resuspended by water currents according to the particle size, unfortunately, comprehensive theoretical predictions of pellet resuspension in the natural environment are difficult and probably unreliable. This will remain so until the dynamics of the transport process are more fully understood or can be expressed in the form of analytical equations (Silvert, 1994). Thus, the establishment of correlations between threshold resuspension velocity and pellet size are essential to modeling of waste dispersion. Because of the difficulties of gathering such data *in situ*, trials were

conducted in a hydraulics laboratory in order to simulate the natural resuspension progress. A range of different sizes of commercial feed pellets (salmon, sea bass and sea bream diets) were first used to determine the threshold current speeds for pellet resuspension and establish any difference in critical current speeds for pellet resuspension between different pellet size to improve waste loading model predictions.

Solid wastes sedimentation

Most studies of the environmental impacts of cage aquaculture have shown increases in the levels of suspended solids and nutrients (ammonia, organic nitrogen and carbon), and decreases in dissolved oxygen levels around cages (Ackefors and Enell, 1990; Weston, 1990; Black *et al.*, 1996). High rates of waste deposition in the sediments below cages can lead to accumulation of organic detritus in the sediments that overwhelm the assimilative capacity of the benthos and result in the formation of anaerobic bacterial mats and anoxic conditions. This in turn leads to the anaerobic generation of hydrogen sulphide and methane. This affects both the fish farm and the environment, with visible changes to sediment quality and recognizable changes in benthic communities in the vicinity of fish farms (Brown *et al.*, 1987; Holmer and Kristensen, 1992; Black *et al.*, 1996). By measuring the organic carbon content and other characteristics of the sediment it is possible to estimate how much wastes have been discharged in the immediate vicinity of the cages, providing baseline data for the dispersion model. However, there are many factors that may influence the accuracy of the post-deposition scenario on the sea bottom near cages including wild fish, bioturbation and resuspension caused by currents (Weston, 1990; Johannessen *et al.*, 1994; Wainright and Hopkinson, 1997; Hakanson *et al.*, 1998). Thus, the deployment of

sediment traps around cages to obtain a more accurate sediment loading is an alternative to modeling waste dispersion.

Sediment traps are a useful tool for collecting solid particles and estimating the quality and quantity of material falling out of the oceanic and limnological water masses (Bloesch and Burns 1980; Blomqvist and Hakanson, 1981; Bloesch, 1994). To complement the investigation of the dynamics of feed pellet transport processes described above, field work on the determination of sedimentation rate at a cage farm site was carried out not only to minimize the assumptions in existing dispersion models but also to validate the existing model. To address the second key question in the present study, field work was carried out on two occasions to derive sedimentation rates *in situ* at a salmon cage farm to allow comparison between measured and modeled outputs as well as to help verify existing models of waste dispersion. The techniques and examples serve to illustrate the merits of this approach to modeling all forms of cage farm wastes in general.

The purpose of the field trials was to allow an indication of sedimentation of various distances from the cages for comparison with those predicted by a dispersion model in terms of C and N loadings. This may be used to help validate this model as detailed production data for the field site was available.

5.2 Materials and methods

5.2.1 Resuspension speed determination in the laboratory

Experiments were conducted in the Hydraulics Laboratory, Department of Civil Engineering, Strathclyde University, Glasgow, UK, during April-July 1999. A flume unit with a channel 3.68 m in length was chosen (Figure 5.1). Laboratory experiments involved constant recirculation of the water within the flume, the flow being generated by a rotating pump. The flume bed was seeded with fine sand (200-600 μm) in a layer 2.5 cm thick and a working section of 2 m in length established. A ruler (length 1 m) was attached outside the flume, with the point of origin being 20 cm upstream of the start of the sediment bed. The water level could be lowered or raised by adjustment of the gate height and current speed controlled by a rotameter. The rotameter consists of a glass tube contained within a steel frame with a scale fixed to one side. Inside the glass tube is a steel float which has been calibrated to rise under known force of water. Water exchange ranged from 5 to 230 L min^{-1}. The rotameter was fed from a submersible pump situated in the tank of the flume connected by flexible hose to a gate valve. The pump was switched on and the valve opened until the required flow rate was observed on the float scale. Opening and closing of the valve raised and lowered the float to the desired levels. Thus, establishing the desired flow rate for the convenience of operation, the rotameter was raised or lowered at interval of 5 L min^{-1} from 10-45 L min^{-1}, to establish a standard curve of the current speed.

The profile of surface and bottom current speed

The flume was established with 25 cm depth of water, approximating 80% maximum depth, and a steady flow rate maintained under the rotameter capacity of 25 L min^{-1}

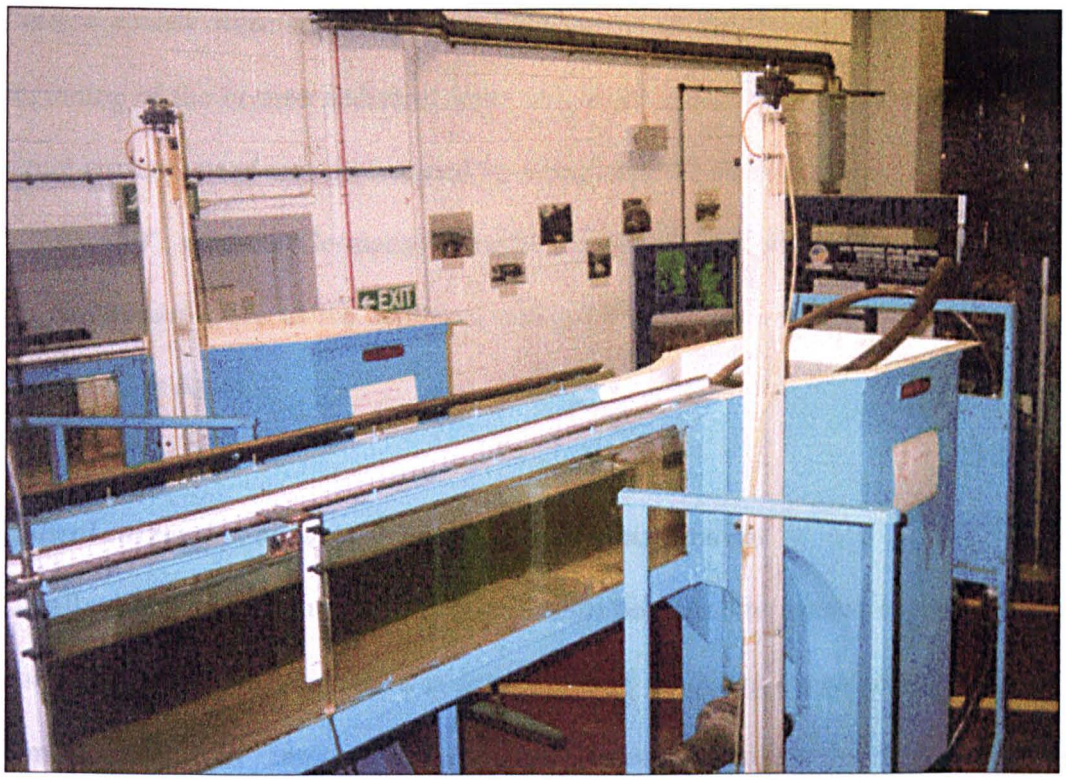


Figure 5.1 Flume unit used in resuspension determination, channel length 3.68 m, channel height 0.32 m, channel width 0.30 m.

which conformed to the critical resuspension speed of most feed pellets described later. Current speeds were measured in the mid-point of the flume, 1 m distant from the beginning of the bottom sediment layer and at the center of the tank cross-section. The actual current speed was determined by using an OTT current meter (A.OTT Kempten, C2"10, 150", made in Germany) close to the surface (~2 cm depth), mid-depth (~12.5 cm depth) and close to the sediment bed (~23 cm depth), with three replicates being made at each depth.

Critical current speed for pellet suspension (resuspension)

Initially, the standard curve of the relationship between the current speed and the input water volume of the rotameter was determined by adjusting the rotameter from 10 to 45 L min^{-1} at 5 L min^{-1} intervals (Figure 5.2). However, because the research focused on bottom current speeds, the water level for the resuspension trials was maintained at 2 cm depth. This was outwith the reliable depth range of the propeller of the OTT current meter. Hence, an alternative way of measuring the current speed was to determine the velocity of 10 replicates of float (plastic ball 20 mm in diameter) travelling across the surface over 1 m length of sediment.

Critical resuspension speed was initially defined as the speed at which 5 out of 10 pellets can be transferred over a 10 cm distance in one minute. When conducting trials, 10 pellets of each type were placed in individual compartments of 24-well multi-dishes containing freshwater in order to pre-soak for 1 min, and the pellets then placed in the flume bed nearly in the same parallel line under static conditions. Only one type of pellet was used each time (two replicates for each diet). Once the pellets was stable in the static water sediment, the current flow was increased with rotameter at 5 L min^{-1}

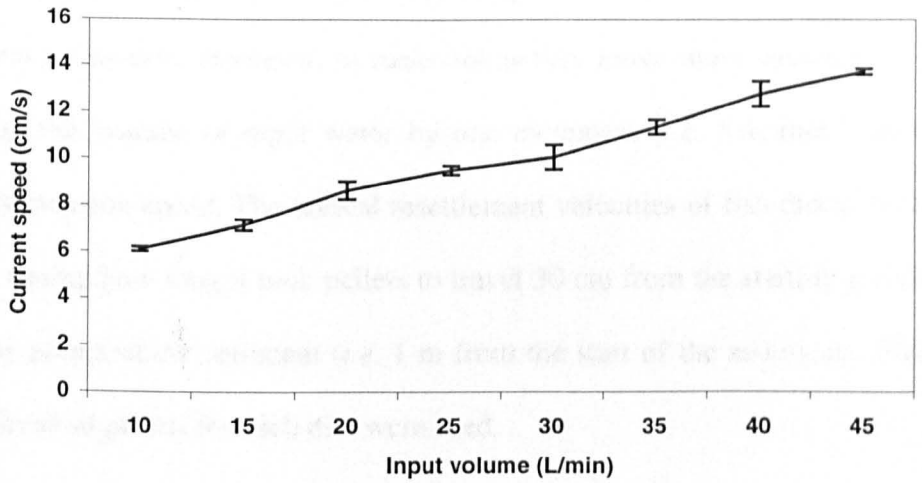


Figure 5.2 The relationship between the input water volume and current speed.

intervals and the pellet numbers that started to move at each current speed noted. The same procedures was followed with all pellets.

Pellet resettlement (moving) speeds determination

The determination of pellet resettlement speed was conducted in the same manner as in previous section. However, to make the pellets move more smoothly, it is essential to raise the volume of input water by one increment (*i.e.* 5 L min^{-1}) above the critical resuspension speed. The critical resettlement velocities of fish diet pellets was recorded by timing how long it took pellets to travel 30 cm from the starting position at the mid-way points of the sediment (*i.e.* 1 m from the start of the sediment). Five replicates of individual pellets for each diet were used.

5.2.2. Determination of in situ sedimentation rates at marine cage farms

General design of the sediment trap system and sampling methods

Sediment traps were designed and made at the Institute of Aquaculture from a design described by Leftley and MacDougall (1991). The trap system was intended to obtain 1) an estimate of the rate of sedimentation and 2) to collect sufficient material for chemical analysis. The trap system was originally designed according to the recommendations made by Wassmann and Heiskanen (1988, in Leftley and MacDougall, 1991) and other researchers (Gardner, 1980a, b; Bloesch and Burns, 1980; Blomqvist and Hakanson, 1981; Hakanson *et al.*, 1989), incorporating the following principles:

- Cylinders with a height:diameter (H:D) ratio > 5:1 should be used. If possible, the H:D ratio should be increased up to 10:1 in unstable water bodies. In the present study the H:D ratio is 7.5:1 (60 cm :8 cm).

- The internal diameter of the cylinders should be larger than 4 cm and between 5-20 cm, or the ratio between trap surface area and trap volume should be < 1, to prevent over-trapping of organic particles.
- It is essential that the mooring maintains the sediment trap cylinders in a stable and vertical position throughout the collection period. Hence, cylinders should be equipped with gimbals in order to maintain vertical position during changes in current velocity and direction.
- Sediment traps should have a freely floating central axis and fins to maintain a constant orientation in the current stream.

Construction

The trap system comprised groups of four cylinders held on a gimbaled stainless steel frame[1]. Details of materials and general dimensions are given in Figure 5.3. The clear container which is screwed onto the cylinder allow sedimentated materials to be readily observed at the time of sample retrieval.

The mooring

The mooring was designed to be inexpensive yet strong and reliable. Pre-stretched polyester rope 10 mm diameter was used for all moorings. These were spliced with PVC eyes and end links to allow connection of buoyancy units and sedimentation units. The sedimentation units were inserted into the mooring line via a 3 m polyester rope finished with stainless steel swivelled snap-shackles so as to allow speedy connection and release during deployment and recovery.

Floatation for each trap units consisted of a 30 cm diameter buoy which gave sufficient buoyancy of the sediment traps at 1.5 m above each trap assembly. The locations of the

[1] Fabrication of the trap units was carried out by Mr. Brian Howie, Institute of Aquaculture, University of Stirling.

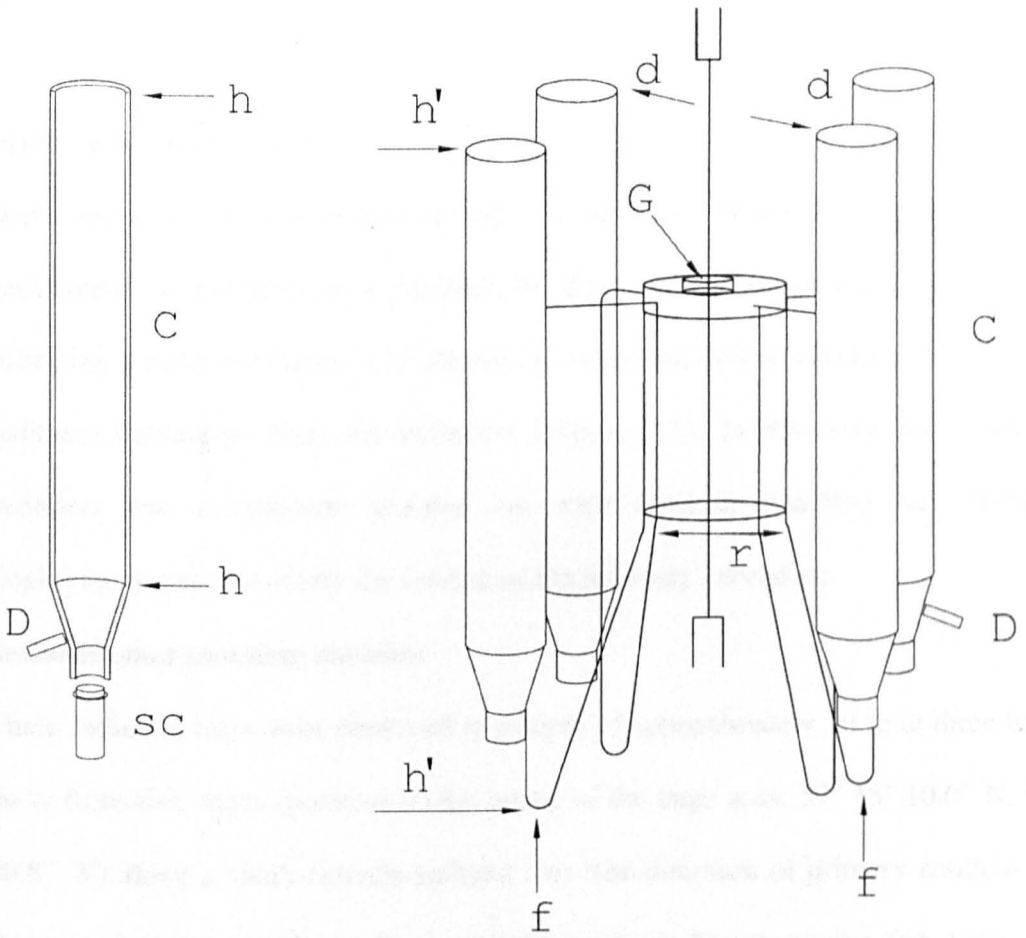


Figure 5.3 An illustration of the sediment trap design. The cross section of the collection cylinder is shown on the left.
C: collection cylinder, D: drain tap, G: gimballed centre rod, SC: sediment container
Distance between points denoted by the same lower case letter are described below.
h: cylinder height (60.0 cm), h': overall height from top of cylinder to the base to the base of the frame (87.0 cm), f: the distance between opposing trap feet (27.0 cm), d: the distance between traps on opposing legs (43.0 cm), r: outside diameter of the support ring (14.2 cm).

mooring were marked on the surface with floats. The mooring was anchored by three 10 kg concrete weights.

Deployment and recovery

Deployment and recovery were carried out manually. When the trap assemblies had been recovered and lifted on to the deck, the drain taps were opened and the water in the collecting cylinders (Figure 5.3) allowed to drain out before carefully unscrewing the sediment containers from the cylinders (Figure 5.3). In this way water within the cylinders was consistently drained out with minimal handling and disturbance. Deployment was essentially the reverse of the recovery procedure.

Sedimentation sampling methods

Three sediment traps were deployed at a depth of approximately 20 m at three locations away from fish cages (position of the centre of the cage area: 57° 15' 10.0" N, 05° 30' 00.8" W) along a south-easterly transect line (the direction of primary residual current flow) at distances of 10 m, 20 m, and 30 m (Loch Duich marine fish farm, Marine Harvest McConnell Ltd., Letterfearn, Kyle, UK). During study, the spring and neap tidal range were 3.38 m and 1.52 m, respectively. Farm production data such as tonnage, feeding rate are given in Section 6.5.3 and Appendix 6.9. The sediment traps were deployed during two experimental periods for lengths of time ranging from three days (3-6 September, 1999) to seven days (6-13 September, 1999) to ensure collection of sufficient and representative samples. The mouths of the traps were located approximately 3 m above the sediment surface. Sedimentation samples were taken by disconnecting the sediment collectors from the bottom of each set of traps. The sediment collectors were brought back to laboratory and the contents poured into appropriate measuring cylinders. After 20 min the sediment samples had stratified and

settled in the measuring cylinders, the upper stratum of water was then carefully poured off to reduce the water volume and the sediment samples were then placed onto pre-weighed foil trays. The sample dry weights were then determined after drying in an oven for 24 h at 105 °C to obtain a stable weight. After drying, each sample was prepared for analysis of carbon and nitrogen content as described in Section 4.2.1. The gross sedimentation rates (GSR), defined as the total amount of material collected in a sediment trap with a known cross-sectional area over a known length of time (Charles *et al.*, 1995; Gremare *et al.*, 1997), was calculated as dry weight (DW) g m^{-2} d^{-1}, while nutrient loadings from sedimentation were expressed as g C m^{-2} d^{-1} and g N m^{-2} d^{-1}.

The feeding rate and FCR information were made available by the fish farmer in order to estimate total waste outputs (see Section 6.5.3).

5.3 Results

5.3.1 The profiles of current speed

The mean current speed at 25 cm depth was determined from readings taken immediately beneath the water surface, in the middle and the near-bottom, and were 9.24, 8.44 and 7.43 cm s^{-1}, respectively (Figure 5.4). From the simple linear regression curve, the true bottom current speed (25 cm depth) under the same pattern was estimated to be 7.29 cm s^{-1} (Figure 5.4).

5.3.2 Critical current speed for pellet suspension

The critical resupension speed for feed pellets was defined as the speed which feed pellets began moving from the static condition. The critical resuspension speeds of all

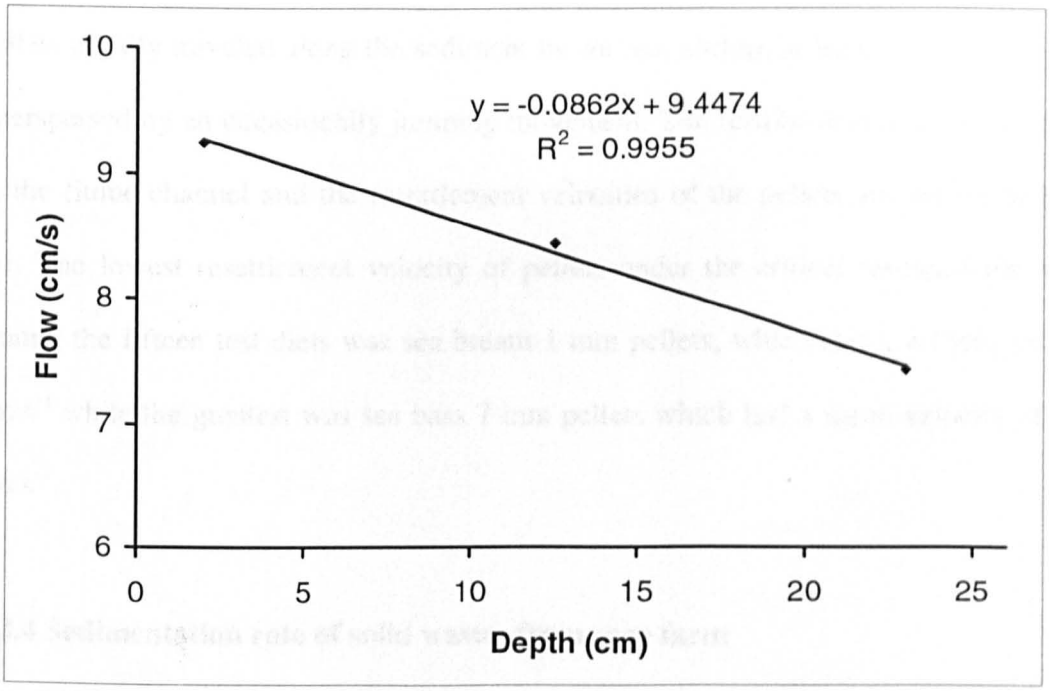


Figure 5.4 The relationship between the current speed at different depths under input water volume of 25 L min $^{-1}$ and 25 cm depth of flume water. The depth was measured from the surface.

pellets, except Trouw 14 mm and sea bream 4.5 mm, was 8.63 cm s^{-1} and stopped at 7.14 cm s^{-1}. The two exceptions began moving at a critical resuspension speed of 9.28 cm s^{-1} and stopped at 8.63 cm s^{-1}.

5.3.3. Pellet resettlement velocities under critical resuspension speeds

When current speeds exceeded the critical resuspension speed, it was found that the fish pellets usually traveled along the sediment by rolling, sliding or hopping along the bed; interspersed by an occasionally jumping movement. The results of *in situ* current speed in the flume channel and the resettlement velocities of the pellets are shown in Table 5.1. The lowest resettlement velocity of pellets under the critical resuspension speed among the fifteen test diets was sea bream 1 mm pellets, which had a velocity of 0.79 cm s^{-1} while the greatest was sea bass 7 mm pellets which had a mean velocity of 3.98 cm s^{-1}.

5.3.4 Sedimentation rate of solid wastes from cage farm

C and N contents of the sediment trap materials are shown in Table 5.2. The estimated sediment nutrient loading rate and the corresponding C/N ratios of sediment surplus are shown in Figure 5.5. C contents ranged from 17.4 to 23.5 % DW. N contents ranged from 1.9 to 2.5 % DW. Average C/N ratios ranged from 7.7 to 9.4, the highest value (9.4) occurring at 10 m station in the first trial.

The temporal and spatial changes in sedimentation rates were between 15.4 and 31.7 g DW m^{-2} d^{-1} in the first trial. Values in the second trial (38.5-65.5 g DW m^{-2} d^{-1}) were twice those in the first trial.

Table 5.1 The critical resuspension speed and the resettlement velocities of different feed pellets.

Diet	Critical resuspension speed ($cm\ s^{-1}$)	Resettlement velocity ($cm\ s^{-1}$; Mean ± S.D.)
Salmon diets		
HE 4 mm	10.09	1.87 ± 0.57
HE 6 mm	10.09	1.91 ± 0.31
HE 8.5 mm	10.09	2.16 ± 0.24
HE 11 mm	10.09	2.41 ± 0.25
Standard 6 mm	9.53	0.85 ± 0.21
Standard 14 mm	10.09	3.46 ± 0.19
Seabream diets		
1mm	9.53	0.79 ± 0.25
3 mm	9.53	1.45 ± 0.16
4.5 mm	10.09	1.69 ± 0.06
5 mm	9.53	1.53 ± 0.20
Seabass diets		
2 mm	9.53	1.84 ± 0.06
3 mm	9.53	2.43 ± 0.26
5 mm	9.53	3.04 ± 0.16
7 mm	9.53	3.98 ± 0.20

Table 5.2 Characteristics of material retrieved from sediment traps at three sampling stations (10, 20, 30 m from cages) on two sampling occasions (A: 3-6 September, 1999; B: 6-13 September, 1999). mean (± S.D.) of sediment characteristics: gross sedimentation rate (GSR; g m^{-2} d^{-1}), mean (± S.D.) total carbon and nitrogen content (mg g^{-1} dry weight) collected from sediment traps.

Sampling occasion and stations	Sediment characteristics		
	GSR	Total carbon	Total nitrogen
A			
10 m	31.7 ± 4.9	235.2 ± 12.5	25.0 ± 1.2
20 m	22.5 ± 0.6	174.0 ± 6.0	20.5 ± 1.9
30 m	15.4 ± 2.1	144.8 ± 12.4	18.7 ± 1.3
B			
10 m	65.5 ± 6.4	216.1 ± 10.0	25.0 ± 1.2
20 m	51.9 ± 2.5	195.1 ± 7.7	23.7 ± 1.0
30 m	38.5 ± 2.0	179.5 ± 6.0	22.2 ± 0.9

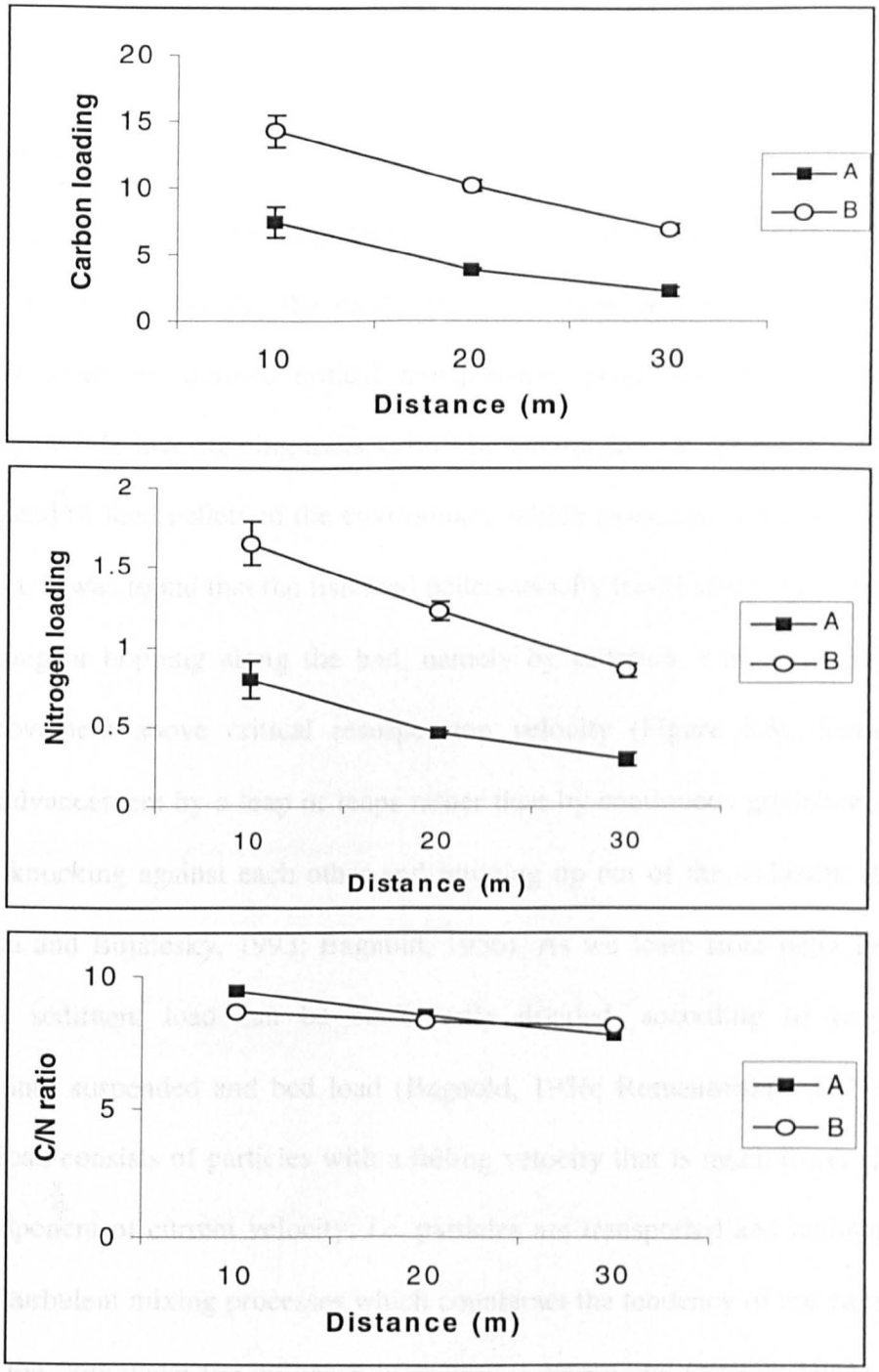


Fig. 5.5 Mean carbon and nitrogen loadings (g C m^{-2} d^{-1}; g N m^{-2} d^{-1}) and corresponding C/N ratios at three sampling stations (10, 20, 30 m) on two sampling occasions (A: 3-6 September, 1999; B: 6-13 September, 1999), estimated from sediment trap data. Error bars represent 1 standard deviation.

5.4 Discussion

The current speeds that cause resuspension of a range of sedimented commercial feeds (1 mm to 14 mm pellets) and the resettlement velocities at which the resuspended pellets move under the defined critical resuspension speed were determined in the present study. While investigating transport of the feed pellets on the seabed, one must study the speed of feed pellets in the environment which associated with current speed and direction. It was found that the fish feed pellets usually travel along the sediment by rolling, sliding or hopping along the bed, namely by saltation, with an occasionally jumping movement above critical resuspension velocity (Figure 5.6). Saltation is defined as advancement by a leap or leaps rather than by continuous gradation such as the gravels knocking against each other and jumping up out of the sediment (Gilbert, 1914, in Isla and Bujalesky, 1993; Bagnold, 1956). As we learn from other bed load movements, sediment load can be customarily divided, according to suspension conditions, into suspended and bed load (Bagnold, 1956; Romanovskiy, 1977). The suspended load consists of particles with a falling velocity that is much lower than the vertical component of current velocity; *i.e.* particles are transported and maintained in the flow by turbulent mixing processes which counteract the tendency of the particles to drop out of the flow under the influence of gravity (Lawson and O'Neill, 1975). While being transported by the current, these particles as a rule rarely touch the bottom, and can cover great distances in suspension. The speed of suspended load is taken to be close or equal to current velocity, whilst bed load moves mainly in the bottom part of the sediment and has a fall velocity equal or close to the vertical component of the current velocity. Therefore bed load is transported by traction or saltation along the bottom, which differs from the movement of suspended load described above.

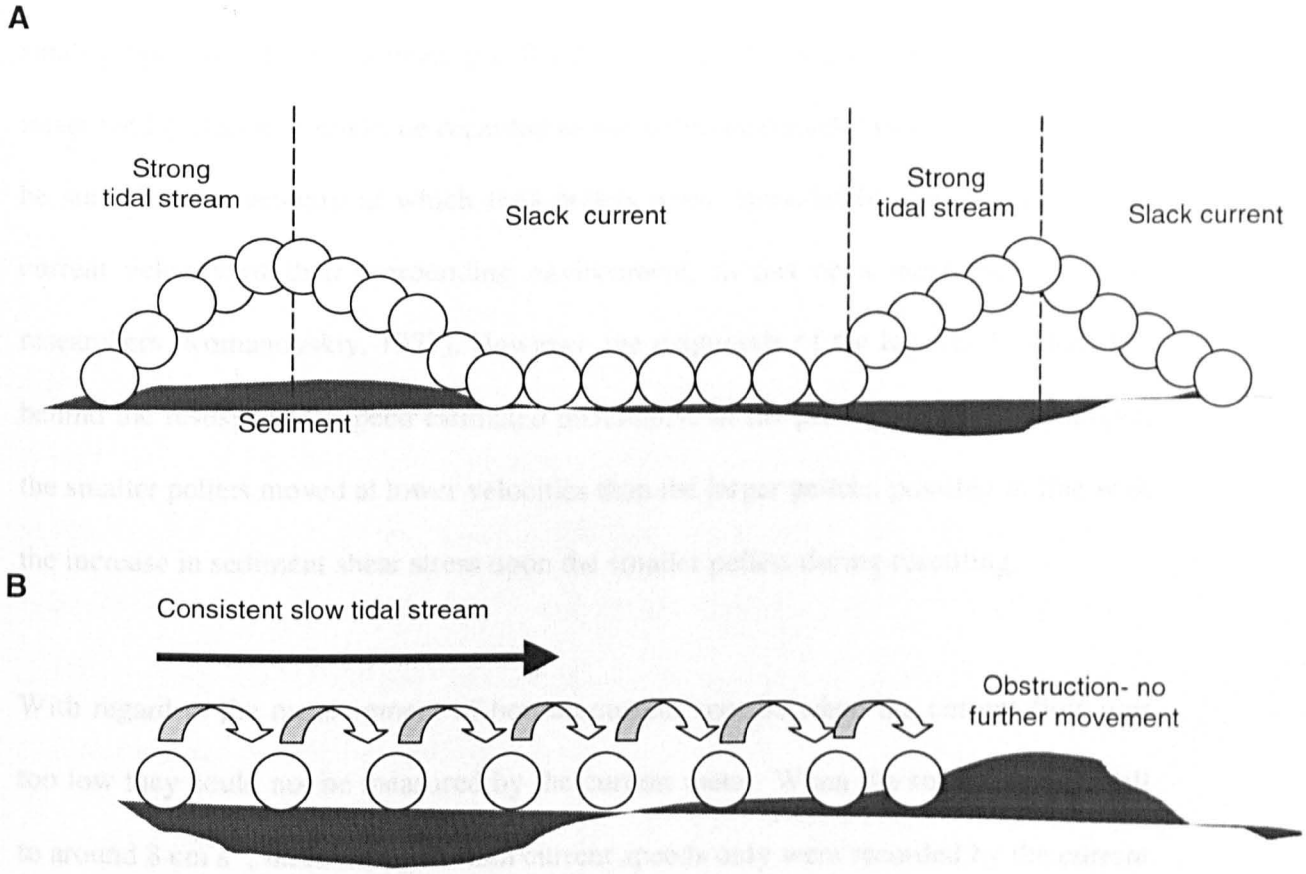


Figure 5.6 Simulated sketch for illustration of particle resuspension (A) and saltation (B).

Experimental observations suggest that feed pellets that detach from the bottom for a period move in the flume should be regarded as bed load, as there is likely to be only a small proportion of fine particles (i.e. 0.2-5.7 % friability, Section 3.3.1) lost from the intact feed pellets that might be regarded as the suspended load. From the study we can be sure that the velocity at which feed pellets move considerably behind that of the current velocity of their surrounding environment, as has been mentioned by other researchers (Romanovskiy, 1977). However, the magnitude of the lag which pellets fell behind the resuspension speed estimated differently. In the present study, for example, the smaller pellets moved at lower velocities than the larger pellets, possibly in line with the increase in sediment shear stress upon the smaller pellets during resettling.

With regard to the measurement of bottom current speeds, when the current flow was too low they could not be measured by the current meter. When the surface speeds fell to around 8 cm s^{-1}, inconsistent bottom current speeds only were recorded by the current meter, because the rotor of the current meter did not spin, or irregularly rotated. It is to be expected that bottom velocities are lower than surface velocities due to the effects of friction as the current flows across the sediment surface. Thus, an alternative method, that of timing the movement of floats over a known distance in the flume tank by stopwatch, was used. It proved straightforward and gave consistent results.

The critical resuspension speed of all diets, which was less than 10 cm s^{-1} in the present study, is contradictory to conclusions by others that resuspension only occurred at high current speeds up to 90 cm s^{-1} (Gowen *et al.*, 1988, 1994).

As the current speed increased, the pellets moved faster along the flume bed. However, it was difficult to precisely determine the critical resuspension speeds. Pellets usually traveled along the bed by rolling, sliding or hopping between the irregularities on the bed of which the fine sand being taken away by the strong current (*e.g.* > 20 cm s^{-1}, pers. obs.).

It is concluded that pellet movement along the bottom takes place mainly by saltation, while the fine particles derived from whole pellets are presumably resuspended. The velocity of the pellets at currents above the critical resuspension (saltation) speed should be taken into account in the waste dispersion model as it may result in greater displacement of solid wastes than predicted by the existing model. It is recommended that further research on saltation or resuspension of feed pellets in relation to different environment bed source is necessary.

A general relationship between sedimentation of material and distance from cages was apparent, *i.e.* more sedimented material was associated with sampling sites closest to the cages. The dry weight of solid wastes was calculated from the total mass collected in the sediment traps. However, it should be noted that this may be an overestimate due to collection of sedimenting planktonic material, seston, and small aquatic animals. The latter were easily removed prior to weight determination.

With similar production data as well as the feeding schemes associated with the two observation periods, it is difficult to understand why there were large differences in sedimentation rate. The amount of wastes produced by a fish farm depends on the total biomass and individual size of fish stocked and the feeding regime. Therefore, the

second trial may have collected more sedimentation material than the first one due to natural factors such as differences in weather or current speed and direction.

The composition of fish faeces depends on the digestibility of the components in food. Data reported in Section 4.3.2 indicate that approximately 30 to 35% of the dry weight of faeces is C, with N contributing about 3%. The C and N content of material collected in the sediment traps in the present study was lower than that of faecal material collected by dissection (Section 4.3.2) or leached faecal material that had been immersed for 2.5 -10 min in sea water (Section 4.3.3). Since the traps were deployed for three or seven days, it is reasonable to assume that the lower nutrient content was due to more extensive nutrient leaching and organic decomposition.

Due to their faster settling velocities most food pellets would have fallen near to the cage, and possibly not have been collected by the traps. It was not possible, due to the nature of the equipment, to position these traps directly beneath the nets where more food would have been collected. Clearly, therefore, there are difficulties in directly quantifying the proportion of uneaten food and faeces which is entering the environment

The C loadings found in the three stations contradict estimates from other findings in temperate areas (Hargrave, 1994), but are similar to those reported for tropical areas (Angel *et al.*,1992). Hargrave (1994) suggested that the threshold for critical C loadings should not exceed assimilative capacity which he estimated to be around 1 g C m^{-2} d^{-1} in marine cage farms in temperate areas. In the present study, the sedimentation rate estimated from Loch Duich site is 1.02 g C m^{-2} d^{-1} (see Sections 6.5.3 and 7.4) which

closely approximates Hargrave's field work in Canada. However, the studies for predicting the environmental capacity of different types of site for cage fish culture are still undergoing and should be used as guides rather than hard and fast rules to development (Hargrave, 1994, Kelly, 1995). Thus, much more work on this area of research is required.

Chapter 6

Application to modelling

6.1 Introduction

Uneaten food and fish faeces are discharged untreated from fish cages and distributed in the environment. The primary effect of settled waste food and faeces from fish cages is the increase of important nutrients, as has been already described in previous sections; *i.e.* total C and the total N content in the sediments. The increase in C and N loadings can lead to an increase in biological activity which, if they are sufficiently high, will eventually give rise to reduction reaction and oxygen depletion in the sediment (Hargrave, 1994; Black *et al.*, 1996; Silvert and Sowles, 1996). This has severe effects on the natural communities and the resultant switch to anaerobic production can cause the evolution of noxious gasses such as hydrogen sulphide. Also, N is generally regarded as the limiting nutrient to phytoplankton production in marine environment and responsible for the development of the algal blooms. Hence, nutrient discharges, especially of C and N from fish farms should be determined simply and as accurately as possible from records of fish production and food conversion ratios (FCR), combined with chemical analyses of feed and fish carcass. Prospective predictions of the inputs (*i.e.* feed) via production and discharges of given chemical elements, on a daily basis and over longer periods of time, would represent a valuable management tool for farmers, and for regulatory and planning authorities.

An existing model is presented here for cage Atlantic salmon farms that integrates biomass, growth rate, FCR, and nutrient retention and discharges in relation to feeding rate, diet composition and key environmental conditions. The mathematical model, originally

described by Gowen *et al.* (1988) for estimating loadings from marine cages, has been further developed by staff at the Institute of Aquaculture (Bostock and Rands unpub. data and Telfer, 1995; hereafter referred to as the 'old model' or 'IOA ver. 1.0') and applied for some years to estimate C input to sediments. This will be modified and validated. The old model (IOA ver. 1.0) was based on a complex multi-layered spreadsheet (data stored in Quattro Pro®), from which sediment loading data outputs in a grid format could be plotted as contours by hand (Telfer, 1995) or exported for use within a GIS environment (Perez *et al.*, submitted).

The basic philosophy of modeling work, described in Chapter 2, has been that the level of sophistication introduced in the models should match what is actually known about processes occurring in nature, and to include in models only those processes that are essential. A flowchart of the modeling processes is shown in Figure 6.1 to clarify description of the operation of the waste dispersion model. It shows the logical pathway through model modules and summarizes the input data required at each step. The new model (*i.e.* IOA ver. 2.0) used here mainly employs Excel™ (Microsoft Inc.) for spreadsheet modelling and Surfer® (ver.6.04, Golden Software Ltd., 1998) for contour map plotting. The Excel model is a multi-layered spreadsheet model in which each page is given a name defined by the data it contains. Surfer is used to produce a contour plot using a grid file output from the spreadsheet and interpolating from the data provided. The conceptual flow-chart illustrates the sequences of steps for operating the model as follows:

1. The production data from the fish farm and food input which are stored in the 'Input Data' sheet indicates the initial source data in the model. 'Input Data' sheet is an important

part of the Excel sub-model where the production data and the basic information for calculating C and N are each entered. All of the Excel spreadsheet cells that are shaded in the model indicate that data must be entered by the user, for example the cage numbers and stocking density, and unmarked cells will be updated automatically (see Appendix 6.1). Information from a lookup table is automatically incorporated based on type of food and fish size.

2. The relationship between settling velocity and feed pellet size is established according to the fish size in the 'look-up table' which is the second sheet of the Excel model. This also includes essential relevant information, such as fish sizes and feed/faeces pellet sizes and feed/faeces settling velocities, for incorporation to 'Input Data' sheet as described above.

3. A mass balance is used to calculate the uneaten amount of food and the faecal material that settle towards the seabed, at rates in part determined by site characteristics and hydrography.

4. The gross carbon or nitrogen output of uneaten food and faeces is stored in two different files (*e.g.* C model and N model), as different formulae are used in their quantification. Before incorporating carbon or nitrogen output into Surfer sub-model, the nutrient leaching rate of faeces can be deducted from the gross carbon or nitrogen output if necessary. The resuspension speed of feed pellets can also be incorporated into the models by deduction of settled material which would be removed when the seabed currents are above a critical resuspension velocity. However, this assumes that any resuspended material will be removed from the locality and thus from the influence of the model.

5. The sum input of food C/N and faeces C/N to sediment are calculated by combining total solid wastes from all cages. The cell coordinates (waste distribution data) of waste output

are in the form of a spreadsheet data, giving spatial sediment concentrations and saved as a grid file, in ASCII format.

6. The grid file is then transferred and entered into a contour plotting program, SURFER® ver 6.04.

7. The position of the cage block can then be superimposed on the dispersion pattern before plotting the waste output contour map from Surfer software.

Input data required by the new model contain two main data sets, *i.e.* production data and *in situ* hydrographical data (*i.e.* current profiles), including the following:

- Annually practical tonnage of fish to be produced at the site
- Food conversion ratio (FCR), based on practical production data at the site
- Depth of water at the site
- Proposed number, layout, size and orientation of cages at the site
- Tidal current measurement throughout the water column over 15 d (spring and neap cycle period), include the speed and direction at depth in site or similar to that of the bottom of the cage, it is possible to average the speed and direction of flow at a particular time point. This would take into account variable water movement and stratification with depth.

Models for each fish farm site should be validated by future monitoring of sediment deposition and modified if necessary based on the additional data. Therefore, three case studies are presented which incorporate new data to explore different aspects of the new model.

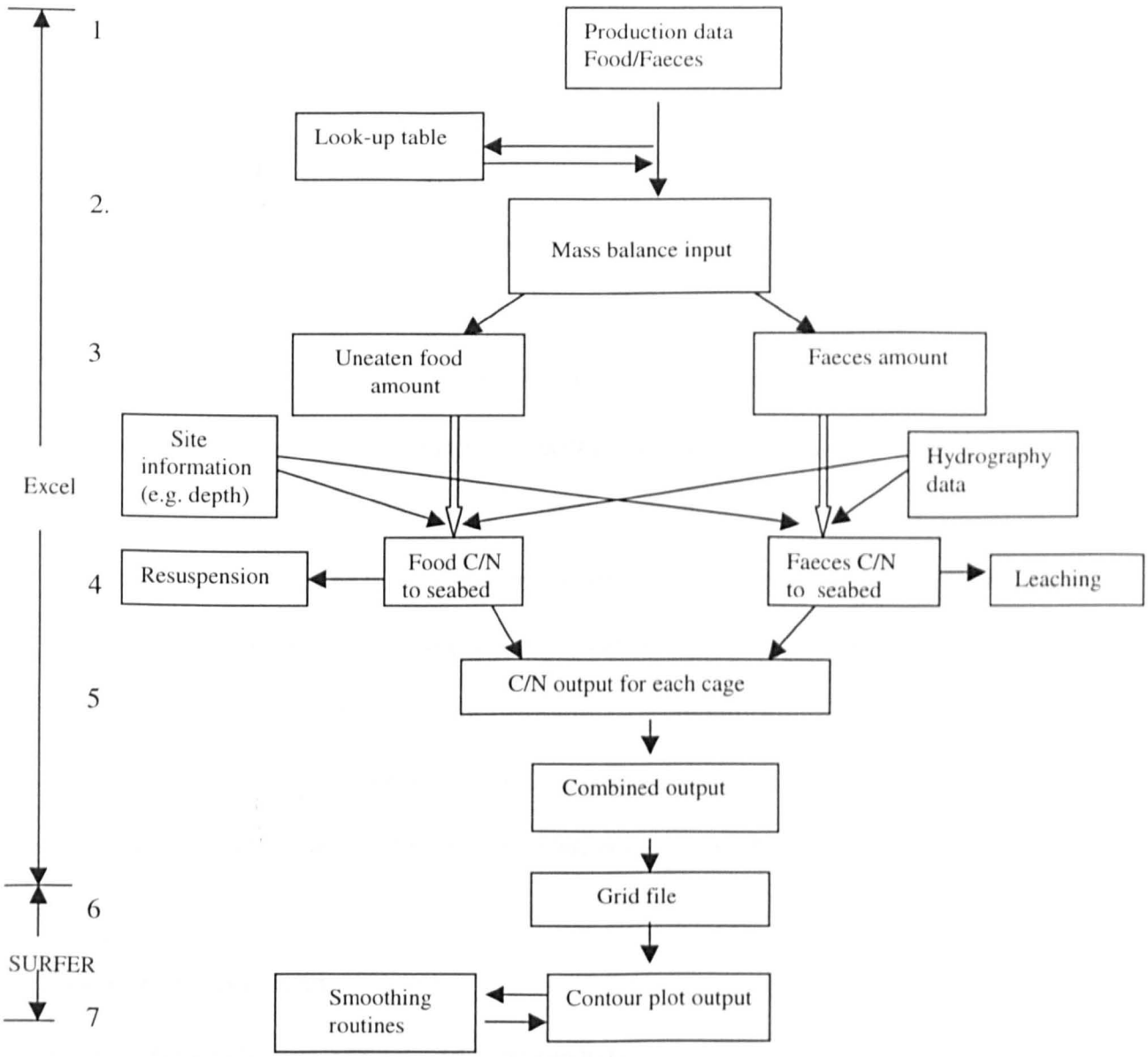


Figure 6.1 Flow diagram of operating the waste dispersion model. Where numbers of procedure refers to text in Section 6.1, ⇳ indicates use of a different mass balance submodel whether C or N is to be calculated.

This chapter has the following four objectives in terms of modelling of wastes dispersion from sea cage farms:

- Review the present knowledge of waste dispersion processes that may be involved in the transfer of wastes from a sea cage to the immediate environment.
- Review existing methods and models currently used for estimating or calculating quantities of wastes and their distribution in order to make recommendations for further development, and to assess how the new model might be further modified.
- Assess the reliability of waste dispersion outputs from the new model.
- Recommend areas for further improvements and identify research needs.

The studies at the focus of this chapter are aimed at modifying the old model with updated data obtained from previous chapters and the validation of the new model by comparison with *in situ* field data collected from fish farms. The modifications include: incorporating current data over a longer period (*i.e.*19 d), settling rates of different pellet sizes, practical production data from cage farms, and the incorporation of nutrient leaching rates.

6.2 The source data

6.2.1 Settling velocities of feed and faecal pellets

In the sheet of the look-up-table (Appendix 6.2), the second column gives the size of the pellets to be fed to the fish of whatever size is shown in the first column. In the third and fourth column of the look-up table, the pellet size is found and the corresponding settling rate of Ewos and Trouw diets under 10 $^{\circ}$C and 33 psu environment is determined using data discussed in Section 3.3. Since settling velocities of some novel pellet types (*i.e.* Ewos 1,2,

3, 12, 14 mm and Trouw 1, 3, 10 mm) were not available, the settling velocities were estimated from a linear regression of the other pellet sizes (*e.g.* Figure 3.6). With regard to the faecal settling rate, the mean faecal settling rate was determined from previous investigation as 5 cm s^{-1} (Section 3.3.7).

6.2.2 Nutrient content analyses

The C and N content of salmon feed pellets, faeces and fish tissues can provide data required to predict the quantity and fate of wastes produced from sea cage farms, and have been determined in previous sections (Chapter 4) for feed and faeces. Smolts (40-80 g fish) were chosen for the determination of nutrient content of fish tissues, due to their ease of homogenisation for analysis of fish flesh content (carcass sample courtesy of Machrihanish Marine Environmental Research Laboratory, Yiogros Toliadis). The results shown in Table 6.1 were calculated in terms of percentage composition dry weight and then converted into percentage composition of wet weight.

Table 6.1 The mean moisture, carbon and nitrogen content of the fish carcass, feed and faeces determined in wet weight.

	Moisture (%)[1]	Carbon (%)[2]	Nitrogen (%)[2]
Fish	73.0	14.3	2.8
Feed	8.0	46.0	7.4
Faeces	80.0	6.0	0.6

[1] mean of three samples; dried at 105 °C for 24 h.

[2] mean of three samples; determined at the same manner as described in Section 4.2.

6.2.3 Current profile analyses

The hydrographical data in Loch Duich were collected as part of a commercial field trial (Telfer, 1998). Data were collected for current speed, current direction and depth, at the Loch Duich site at 20 min intervals for 19 days (Appendix 6.3). The current profile data were also stored in the first sheet 'Input Data' of the spreadsheet model. Current data was collected and used in the spreadsheet model according to the following conditions:

1. Collect a series of continuous current data for at least 15 d to sample half a lunar cycle over the complete spring and neap tide range.
2. Sort current data into eight compass bearing vectors, (N, NE, E, ES, S, WS, W, WN) into a table format (see also Appendix 6.3).
3. Assign a proportion of the current data to each of the current vectors according to the number of readings.
4. Calculate faeces and food loadings for each vector by multiplying the total C and N loading for faeces and food, respectively, by the proportion assigned to that vector.
5. For each vector calculate the mean current speed and standard deviation.

6.3 Mass balance model

The relative carbon and nitrogen loadings from waste feed and faeces at the site were calculated using mass balance equations (Gowen *et al.*, 1988) as illustrated in Figure 6.2. based on the updated data from Table 6.1. The data was stored and shown in the first sheet of the spreadsheet model (*i.e.* 'Input Data' sheet; see Appendices 6.5 and 6.8). The mass balance was calculated using the following data and assumptions:

1. Waste feed input estimated at 10% (Hargrave, 1994).
2. Total C and N content of feed estimated at 46% and 7.4%, respectively (Table 6.1).
3. C and N content of fish are 14.3% and 2.8%, respectively as determined directly from the fish tissues (Table 6.1).
4. 50% of consumed carbon is respired (Gowen *et al.*, 1988) and 39-53 % of consumed nitrogen is excreted as ammonia (Figure 6.2).

Although variations in the amount of feed wasted as a result of farming practice are difficult to quantify, the amount of uneaten food is an important factor in determining the degree of benthic enrichment. The estimated 5-10% waste feed input at cage farms was derived as follows. In Scotland most salmon farmers use commercial pelleted food and achieve an FCR of between 1.0 and 1.2 (SOAEFD, 1996). The work of Juell (1991), Blyth *et al.* (1993), Findlay and Watling (1994), Hargrave (1994) and Foster *et al.*(1995), which detected uneaten food pellets by acoustic sensors for this range of FCRs, indicate that a value of 5-10% is a reasonable estimate for food wastage.

Faecal waste input was calculated according to the following principles. Because the modeling of waste dispersion is focused on solid wastes, the C and N solid wastes can be simplified as follow:

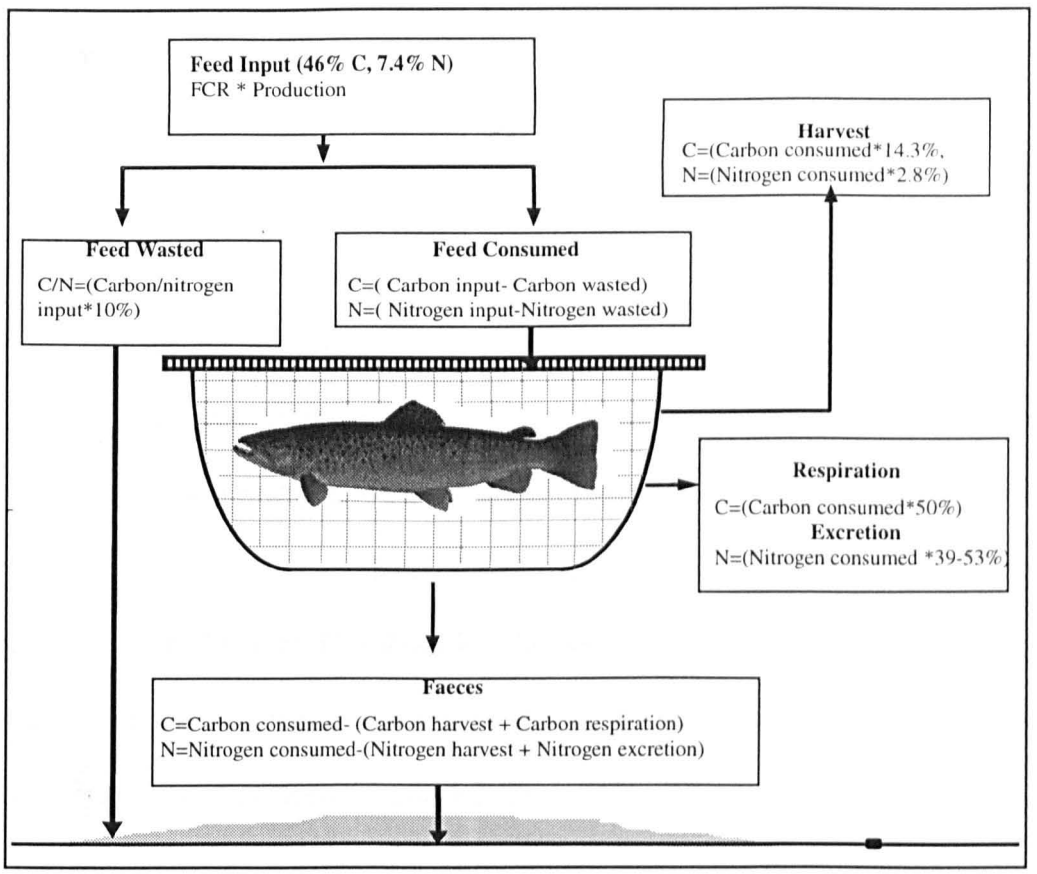


Figure 6.2 The mass balance model to calculate the carbon and nitrogen wasted from uneaten food and faecal material in salmon cage farms (modified from Perez *et al.*, submitted)

C waste = C in faeces + C in waste feed (6.1)

N waste = N in faeces + N in waste feed (6.2)

However, quantification of the amount of faecal wastes is difficult as it will vary with feed composition and fish species. By using a mass balance approach, the faecal C waste was estimated from the total feed input by deducting the C harvested and that lost by fish respiration, whilst the faecal nitrogen waste was calculated by deducting the harvest and other outputs (*e.g.* ammonia excretion). Thus, the average flux of C and N through a salmon cage farm were as presented below. This describes the nutrient output from a cage-fish farm without considering the environmental parameters. Hence, the equations for the waste loads of C and N converting from equations 6.1 and 6.2 respectively are:

C waste = (Total C consumed – harvested – respired) + C in waste feed (6.3)

N waste = (Total N consumed – harvested – other output (*e.g.* ammonia)) + N in waste feed (6.4)

The quantities of wastes released to the environment, particularly total carbon and nitrogen in the form of uneaten food and faecal wastes, are calculated using a mass balance equation as described above and based on the data from published figures or from the figures shown above (Table 6.1). For example, the carbon content in fish food and fish in the present study was 46% and 14.3%, respectively (Table 6.1). Thus, one tonne of food contained 460 kg of carbon and one tonne of salmon contained 143 kg of carbon. Assuming a food conversion

ratio of 1:1 and wastage of 10%, to produce one tonne of fish 414 kg of carbon are consumed but only 143 kg carbon are retained by the fish, *i.e.* 35% of the consumed carbon is retained by the fish and 65% lost as faecal and respiratory carbon (*i.e.* including 50 % was attributed to respiration). The composition of the faeces and urine obviously depends on the digestibility of the components in the food. Feed protein content also influences the composition of the urine (Rychly, 1980). Previous studies (Chapter 4) indicated that approximately 30-40% of the dry weight of faeces was C whilst N contributed about 2.8-3.5% of the dry weight. Alternatively, the amount of faecal nitrogen waste can be determined from the digestibility of protein and the protein content of fish food. For example, the nitrogen content in fish food and fish in the present study was about 7.4% and 2.8%, respectively. Thus, one tonne of food contained 74 kg of nitrogen and one tonne of salmon contained 28 kg of nitrogen. Assuming a food conversion ratio of 1:1 and wastage of 10%, to produce one tonne of fish then 66.6 kg of nitrogen were consumed but only 28 kg nitrogen were retained by the fish. That is 42% of the consumed nitrogen was retained by the fish and 58% lost as faecal and excretory nitrogen. The values are in good agreement with the estimates of Gowen *et al.* (1988) and Talbot and Hole (1994).

6.4 Waste dispersion model

The model, developed by Bostock and Rands, was based upon the equation of Gowen *et al.* (1988) for the horizontal dispersion of uneaten food and faeces and is described in Telfer (1995, see Figure 2.2).

This equation made the following assumptions:

1. There was no resuspension or re-distribution of the waste after initial settling on the seabed.
2. The current speed and direction was uniform with depth.
3. The depth throughout the dispersion area was constant.
4. Food and faecal pellets did not disintegrate during the settling process.
5. Density of feed and faecal particles remained the same during settling.

Loading and dispersal from each cage was calculated. Dispersion from the cages was determined at 10 m intervals for a distance of 200 m along eight compass axes (N, NE, E, SE, S, SW, W and NW). The degree of faecal waste dispersion at each settling distance was calculated from the current speed which was assumed to follow a normal distribution, with the loading at each distance being weighted by its relative probability, using the equation:

$$Z = \frac{I}{\sigma\sqrt{2\pi}} e^{1/2((Y-u)/\sigma)^2} \qquad (6.5)$$

Where:

Z is relative probability (%)

σ is sample standard deviation

Y is distance (m)

μ is sample mean (m)

The faecal wastes distribution and current distribution were combined using Monte Carlo analysis to produce the resulting distribution especially in filling gaps (based upon normally

distributed data) upon which the settling distances, sample mean and sample standard deviation are calculated.

Waste output was in a numerical grid format, with each square representing 100 m^2 area (*e.g.* Appendix 6.4). A contour plot was then formulated.

Using both numerical data and contour plots the direction and quantities of waste and waste dispersion within the sediments was estimated. This demonstrated the effectiveness of the tidal currents at the site to disperse waste or where problems of accumulation may occur in the future. The area over which loading occurs at each 10 m interval along the axes was calculated assuming that the loading covered the 10 m segment between these intervals. The proportion of cage loading along each current vector was calculated from the amount of current flow in that segment, with the initial input area being 25 m^2 at the centre of each cage.

6.5 Application to modelling

Three case studies were used to investigate the present mass balance model. The first study provided growth data on Atlantic salmon reared in indoor tanks (from 0.1 kg juvenile smolt grew to 2.1 kg adult salmon) and fed a commercial artificial diet. Data were provided by Dr. William Roy (Marine Environment Research Laboratory in Machrihanish, IoA, University of Stirling). The second data set came from a salmon sea cage farm, Aquascot Ltd (Mheall Bay, Orkney, UK) and was provided by Dr. Derek Roberston (External Facility Director, IoA, University of Stirling). The third was provided by Mr. Colin Tulip, manager

of Loch Duich marine fish farm, Marine Harvest McConnell Ltd., Letterfearn, Kyle, UK (same location as field work described in Section 5.2).

Although there are three sets of production data, the hydrographic data that are essential for waste dispersion modeling are only available from the last case study. Hence the final contour plot part of the dispersion model will be carried out for the Loch Duich site only.

6.5.1 Trial 1 : Machrihanish indoor tanks data

The production figures provided by the Marine Research Laboratory in Marchrihanish were incorporated into the mass balance equations as shown in Appendix 6.5. The mean initial fish size was 0.11 kg on 15 March 1997 and reached 2.05 kg after one year on 15 March 1998 with a food conversion ratio (FCR) of 1.28. As described in Section 3.1, most reported loss due to uneaten food have been reduced to 5-10% of the total food input. To enable easy comparison, the proportion of uneaten food from three case studies was thus assigned by two scenarios, *i.e.* 5% and 10 %. And C and N composition of fish feed and fish flesh in three trials were assumed to be the same in the three trials (*c.f.* small variability of feed content in Tables 4.2 and 4.3) for the sake of convenient comparison.

Using the above production value in the cage cultivation module and the mass balance model, and assuming the initial stocking density of smolts to be 5 kg m^{-3}, a 15 m x 15 m x 10 m cage would initially have held a maximum of 11,250 smolts of fish size 0.11 kg in March 1997. The overall survival rate in smolt year classes put into sea was estimated

around 90% (SOAEFD, 1996). The highest mortality of cultured fish was assumed to be 5% in the first month due to what is termed by the industry as 'failing smolt syndrome'. These smolts fail to survive after transfer to sea for a variety of reasons, principally that the smolts are unable to adapt to the marine environment. However, the mortality assumed to be 0.5% per month in subsequent months was in good agreement with the overall survival rate which was in excess of 90%. The mean monthly feeding rate was varied according to the water temperature and the fish biomass at the site. Thus, the total biomass, FCR, feed input and weight gain could be calculated (Appendix 6.5).

The carbon and nitrogen contributed by uneaten food and fish faeces to the environment in the vicinity of the cages was estimated separately in the spreadsheet. If the value of 5% as estimated above was assumed to be the uneaten food proportion, the total loadings of carbon and nitrogen from a typical tank culture fish farm was estimated to be 166 and 28 kg, respectively, per tonne of fish produced per year (Table 6.2, also see Appendices 6.6 and 6.7 for detail). If 10% uneaten food being taken into the model the solid waste of carbon and nitrogen accounted for 181 and 34 kg, respectively, per tonne of fish produced per year (Table 6.2).

6.5.2 Trial 2 : Aquascot Ltd. cage farm data

The mean initial size of fish was 0.06 kg on 31 January 1997, reaching 3.15 kg after one year on 31 January 1998 with an FCR of 1.11 (Appendix 6.8). Using the information given

Table 6.2 Summary of waste outputs from three case studies applying the present mass balance model and using two scenarios of uneaten food 5% and 10 %, respectively. Fish sizes and FCR were provided by fish farm manager or calculated from practical production data (see Sections 6.5.1-6.5.3).

Site	Fish size (kg)	FCR	Uneaten food (%)	Total C waste (kg)[2]	Total N waste (kg)[3]
MERL[1]	0.11-2.05	1.28	5	166	28
(Machrihanish)			10	181	34
Aquascot Ltd.	0.06-3.15	1.11	5	125	24
(Orkney)			10	138	29
Marine Harvest	0.97-1.43	0.98	5	94	21
(Loch Duich)			10	105	25

[1]Marine Environmental Research Laboratory, Machrihanish, IOA, University of Stirling.
[2]Total C waste = C in waste feed + C in faeces, units are kg C per tonne of fish produced per year.
[3]Total N waste = N in waste feed + N in faeces, units are kg N per tonne of fish produced per year.

above, the proportion of 5% and 10% of food input were considered as uneaten food. Thus, the total loadings of C and N from a typical cage-fish farm was estimated to be 125 kg and 24 kg for 5% uneaten food and 138 kg and 29 kg for 10% uneaten food, respectively, per tonne of fish produced per year (Table 6.2).

6.5.3 Trial 3 : Loch Duich cage site

The carbon dispersion predicted from modeling at the Loch Duich site was based on a 10 d operational period with a production biomass of 114.5 t fish and an FCR value of 0.98, data being provided by the farm manager at the site (Appendix 6.9). The farm was composed of 24 cages, only five of which were in operation at the time of sedimentation sampling. From the mass balance calculation, 1,247 kg of solid carbon were exported as waste during the 10 d period of feeding, 537 kg and 710 kg due to uneaten food and faeces respectively (Appendix 6.9). It was also assumed 5% and 10% of food input being considered as uneaten food. From the mass balance calculations the total loadings of C and N was estimated to be 94 kg and 21 kg for 5% uneaten food and 105 kg and 25 kg for 10% uneaten food, respectively, per tonne of fish produced per year (Table 6.2).

6.5.4 Contour map output (Trial 3 : Loch Duich cage site)

It was assumed 10% of food input being considered as uneaten food at the site. From the mass balance calculations as described in Section 6.5.3, 1,246,898 g of solid carbon were exported as waste during the 10 d period of feeding, 537,026 g and 709,872 g due to uneaten food and faeces respectively (Appendix 6.9). The contour maps of the predicted dispersion and loadings of total C and N are shown in Figure 6.3 and 6.4. The C and N was

distributed mainly to the north-west and south-east of the site, along the main direction of current flow. The highest concentrations were located under the cage, reaching values of 3,089 g C m^{-2} and 546 g N m^{-2} over the 10 d period, respectively.

6.5.5 Nutrient leaching

From previous investigations (Section 4.3), the leaching rate of feed pellets is negligible during sedimentation considering the short time taken for the feed pellets to reach the seabed. However, a rapid loss of faecal nutrients occurred 2.5-10 min after immersion in sea water, *i.e.* total C and total N were found to leach by as much as 22 % and 26%, respectively, after 5 min immersion on one sampling occasion (see Section 4.3). In order to incorporate the leaching loss from faecal waste into modeling, nutrient leaching values of 20% C and 25% N from faecal waste, respectively, are deducted from the mass balance model outputs. Thus, a new contour map (Figure 6.5) showed the highest concentration of organic carbon loading reduced from 3,089 g C m^{-2} to 2,965 g C m^{-2}, also resulted in a 12% reduction in total (feed+faecal) C waste. The highest concentration of organic nitrogen loading fell from 546 g N m^{-2} to 479 g N m^{-2}, also resulted in a 16% reduction in total (feed+faecal) N waste.

6.6 Preliminary comparison of measured and modelled results

A preliminary comparison between actual and modelled sedimentation may be undertaken using the sedimentation rate determined in Section 5.3.4 and modelled waste inputs from the contour map given in the previous section. This will allow an estimate of the performance of the model for estimation of field conditions at the Loch Duich cage site.

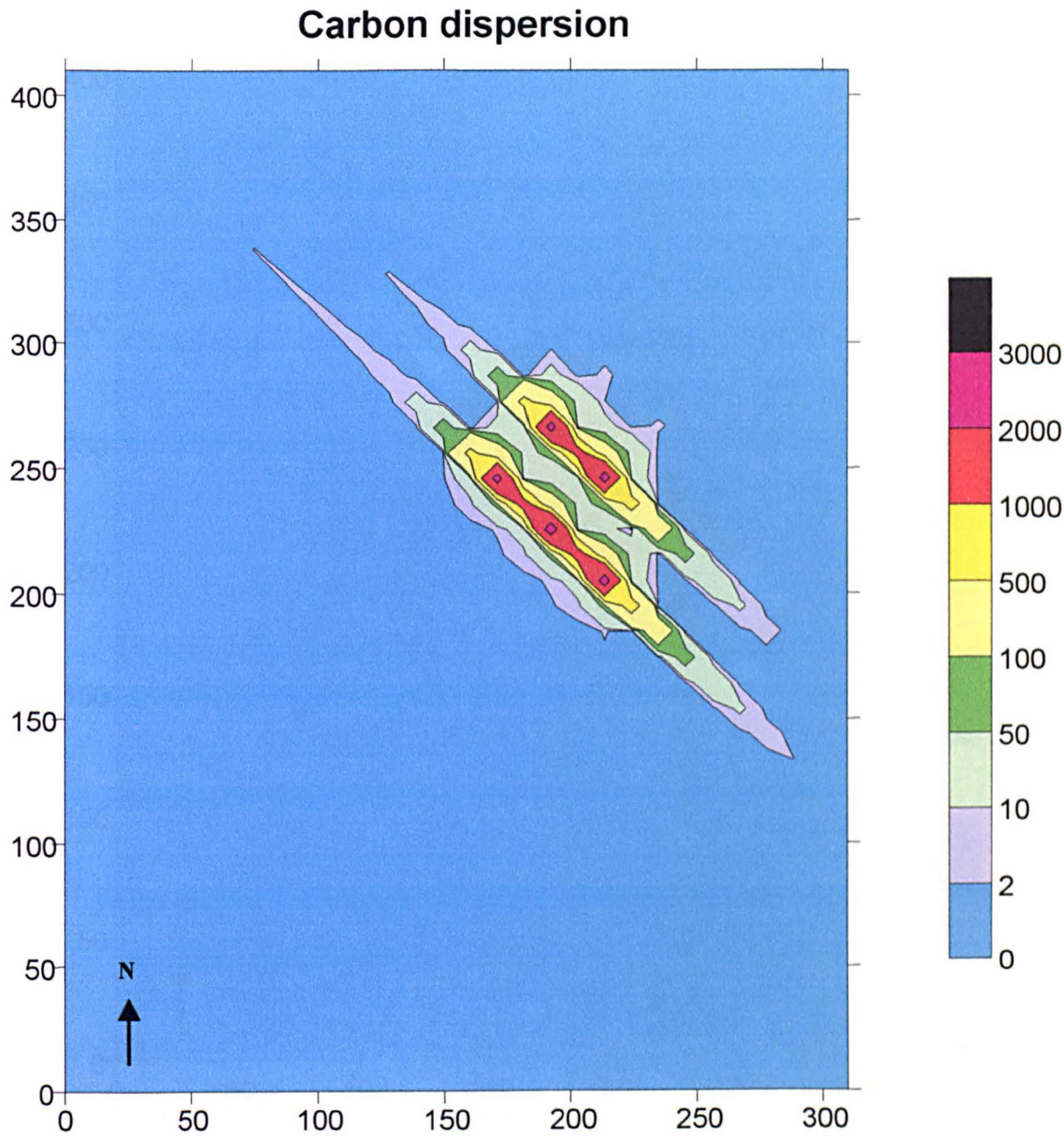


Figure 6.3 Predicted dispersion and benthic loadings of total carbon over a 10 day period at Loch Duich sea cage site. Concentration units are g C m^{-2}, axis units are metres.

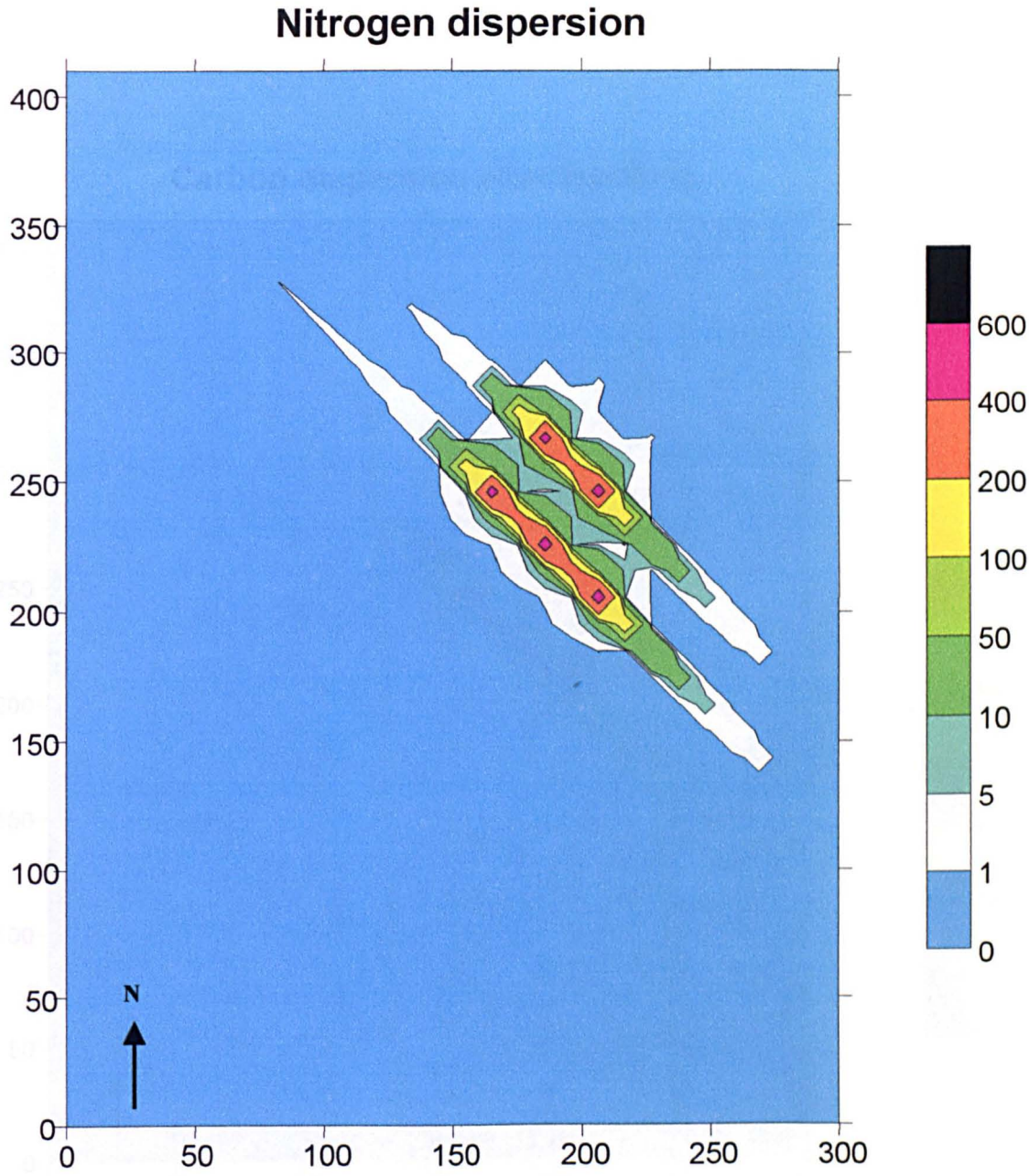


Figure 6.4 Predicted dispersion and benthic loadings of total nitrogen over a 10 day period at Loch Duich sea cage site. Concentration units are g N m^{-2}, axis units are metres.

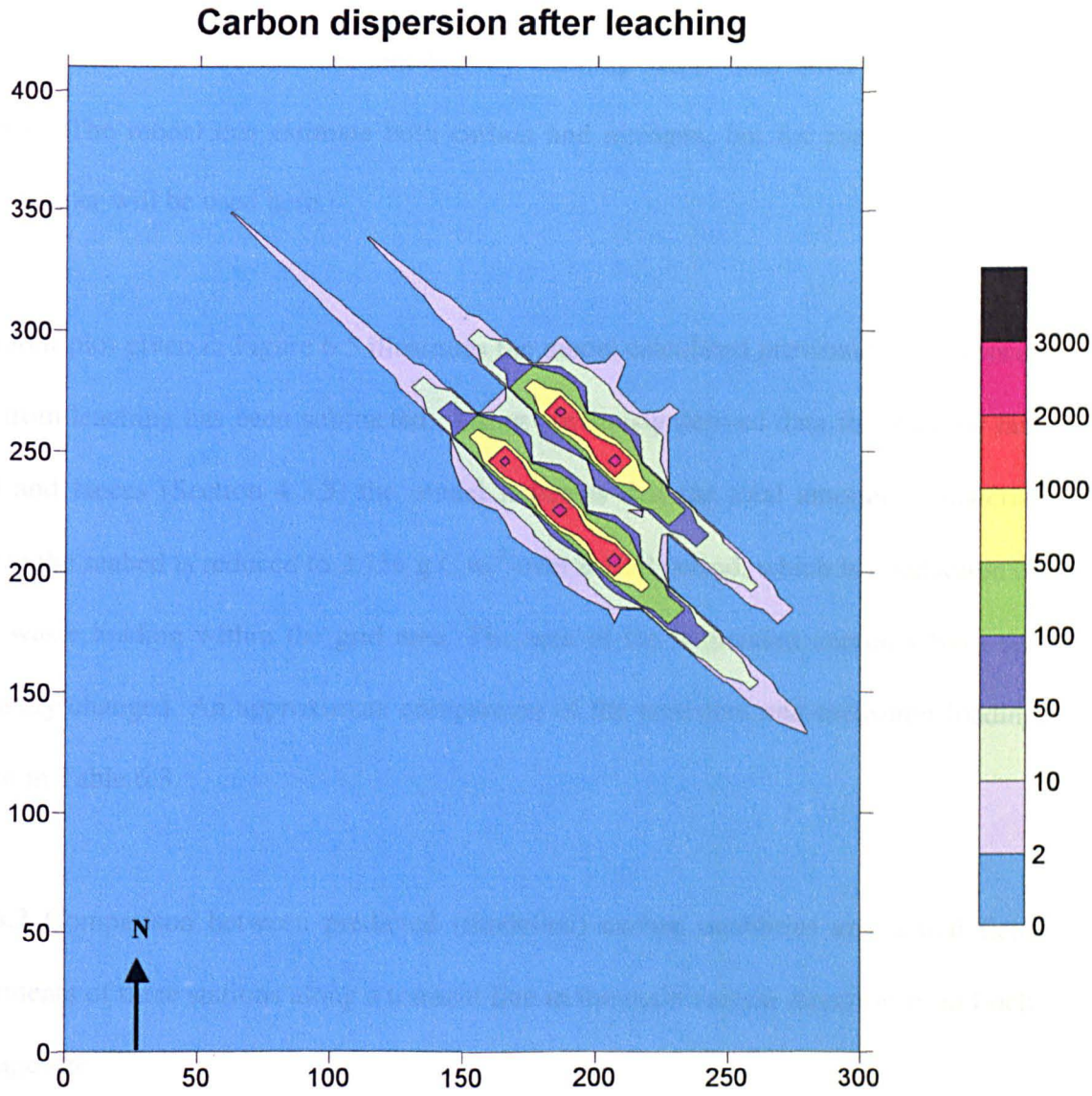


Figure 6.5 Predicted dispersion and benthic loadings of total carbon incorporation leaching data over a 10 day period at Loch Duich sea cage site. Concentration units are g C m^{-2}, axis units are metres.

Material collected in the sediment traps are representative of the amounts of material settling to the sea bed with nutrients only being lost due to leaching. Resuspension and saltation after settlement cannot be calculated from this data. Therefore the model used in comparison will only take into account loss by leaching rather than other environmental parameters. The model can estimate both carbon and nitrogen, but for convenience sake only the former will be used here.

The contour plot given in Figure 6.5 illustrates the model calculated previously after loss of carbon from leaching has been subtracted. Using previously derived data for leaching rate of food and faeces (Section 4.3.3) the model indicates that the total amount of material settling to the seabed is reduced to 2,756 g C m^{-2} over a 10 d period, which is a reduction of 11% in waste loading within the grid area. The area of the dispersion contours have not significantly changed. An approximate comparison of the predicted and measured loading are given in Table 6.3.

Table 6.3 Comparison between predicted (modelled) carbon quantities and actual field measurements of three stations along a transect line in the main current direction from Loch Duich cage site.

Distance from the cages (m)	Modelled nutrient (g Cm^{-2})	Measured nutrient (g C m^{-2})
10	500-1000	75 – 143
20	80-500	39 - 101
30	10-80	22 - 69

Although it is difficult directly comparing the contoured output from the model with the measured amounts of carbon loading, the results show a general agreement between the sedimentation input from stations at 20 and 30 m but not at 10 m from the cages (Table 6.3). This shows the model overestimates the loading near to the cage block, probably because it does not allow for post-settlement removal of material by resuspension processes.

Full model validation will require a longer term investigation using sediment trap data and seabed sampling at several sites of varying hydrographic regimes. However, the present investigation suggests that this model shows a reasonable comparison with field conditions at distances at 20 to 30 m from the cages. This is important as field survey data indicates that the transition zone between impact and background conditions often occurs within this range of distances (Gowen *et al.* 1988; Black, *et al.*, 1996; Harvey and Phillips, 1996; Institute of Aquaculture, unpublished data).

6.7 Discussion

The waste dispersion model presented here has eliminated a number of assumptions (*e.g.* single pellet settling velocity, no leaching during sedimentation) from previously used modeling techniques. Also, minimal data inputs are needed to run the model, reducing the cost of field surveys. The model predicted a smooth negative gradient in waste inputs to the sediments as the distance from the cage increased, in agreement with the work of Hevia *et al.* (1996). Although the model performance is acceptable and the model predictions largely

are in agreement with field data from Loch Duich, further validation of the model in different scenarios is required to test if predictions are consistently good. From previous investigation of critical saltation velocities for feed pellets in Section 5.3, it was found that the velocity at which feed pellets move considerably behind that of the current velocity of their surrounding environment due to the shear stress of sediment upon the feed pellets. To incorporate the post-deposition loss of waste particles caused by resuspension/saltation after settling on the sea bed, the velocity of the pellets at currents above the critical reuspension/saltation speed should be taken into account in the waste dispersion model. This will result in a greater dispersion footprint of solid wastes than those predicted by the existing model. In future, this data for feed pellets may be incorporated into the model once the complex mathematical relationship for tracking particles in the water column after resuspension are established. However, the faecal resuspension velocity can only be taken into account when that information becomes available in the future.

Simple models that predict the dispersion and inputs of organic waste from cage fish farms can be used as management tools for the siting of farms and assessing potential enrichment of the benthic ecosystem. The model developed here predicts the amount of organic C and N originating from uneaten food and fish faeces accumulating in the near vicinity of a marine cage farm in terms of mass of C and N g m^{-2} d^{-1}. The model can either be used prior to new site development in order to predict waste loading or estimate additional total loading if a site is to change fish production. Furthermore, the model can be used for all forms of cage aquaculture provided the production data, the current data at the site and pellet settling velocities under various environmental conditions are available.

All models are based on assumptions that arise from both a lack of knowledge of the processes involved and a subjective assessment of their importance. Assumptions normally have a degree of uncertainty and inaccuracy, so it is necessary to reduce their number to the minimum in order to increase the accuracy of predictions. One assumption of the old model is that the nutrient content of faeces does not change during sedimentation. The updated version of the model for dispersion of solid wastes that takes account of losses by leaching, reducing overestimates of nutrient inputs to the sediment. However, there may be important inputs of dissolved nutrients to the water column that may result in the enhancement of eutrophication at cage sites. Although the new model introduces nutrient leaching of waste pellets, it is likely that this will vary from site to site. One important assumption remaining in the new model is the proportion of food going to waste as uneaten. Especially in Trial 1 (Machrihanish site), 5% of uneaten food should be more reasonable and this was in good agreement with the views of staff at the site who try to minimize uneaten feed in the trial and who estimated that uneaten food was probably less than 5% of food fed (Dr. William Roy, pers. comm.). In the cage site, some of this will be consumed by wild fish or benthic animals, although the amount consumption is difficult to quantify. If it is assumed the proportion of wasted feed consumed by wild animals is around 50%, for example, as has been reported at some sites (Smith *et al.*, 1997; Dougall and Black, 1999), then the proportion of the total food given to the fish that falls to the sediments is only 5% and 2.5% for the two scenarios respectively, which represents a potential large reduction in the waste loadings. Therefore the model may require a degree of site-specific validation.

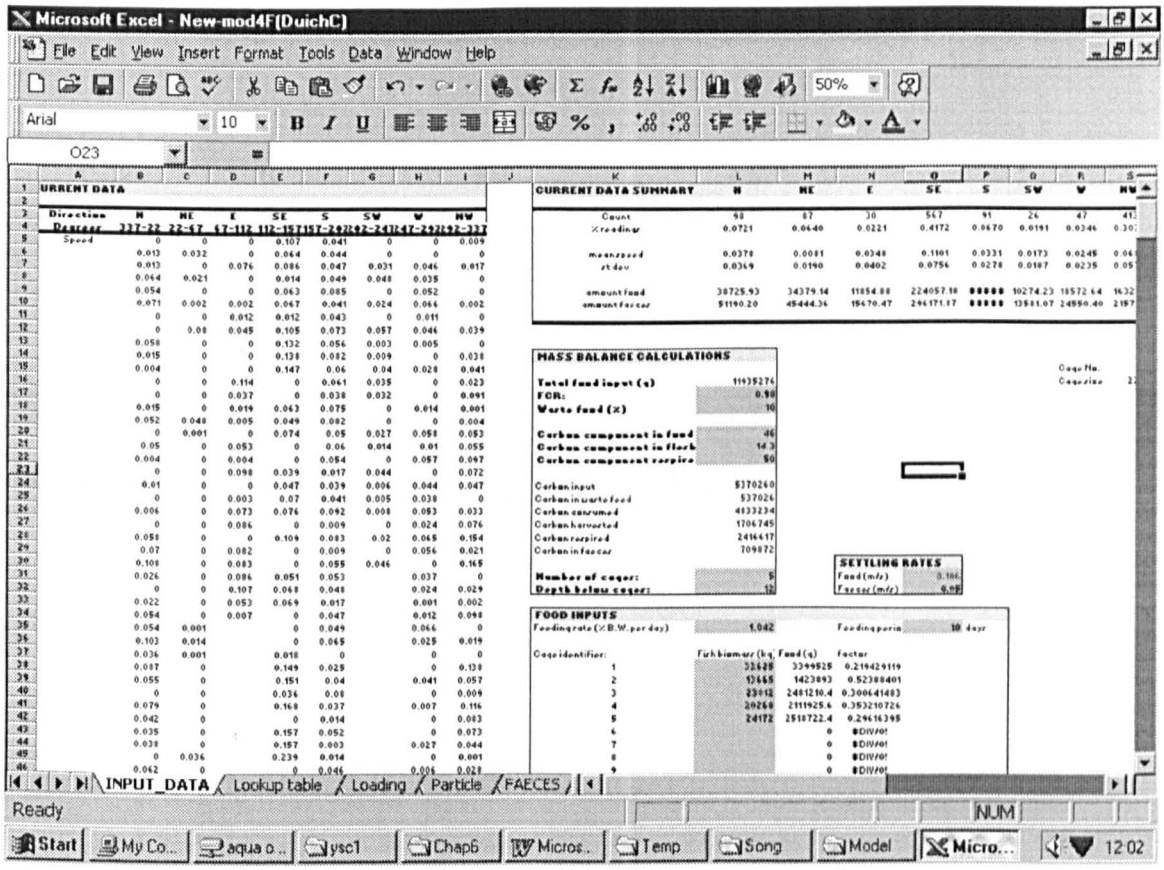
The dispersion model using here assumed that the water current was constant throughout the water column. However, water velocities within the cage itself can be reduced by 20-40% (Inoue, 1972), which would result in greater accumulation beneath the cage and less dispersal. The current data in the present study is sorted into eight compass bearing segments into a table format. Though the combined widths of these segments account for 360 degrees a better method to apportion the distribution of wastes would be according to each individual current vector.

The choice of an appropriate model for solid wastes dispersion will depend on its waste origin, its reactivity with various particulate matter, and on the purpose of its application. Models should always be evaluated in terms of the sensitivity of their results to the importance of neglected processes (GESAMP, 1991a). Hence, the choice of data sets for model 'tuning' or verification must be done with considerable care. An alternative modeling approach is also under development at the Institute of Aquaculture using a GIS and this can be used to compare the output from the new model presented here. The powerful geographic modeling functions of the GIS-based system should increase the accuracy of the dispersion and loading predictions as more grid information can be provided. Initial trials using the IDRISI GIS package has given encouraging results (Perez *et al.*, submitted).

The contour map of the C and N waste dispersion from the new model (IOA ver. 2.0) that takes account of both type of food (*i.e.* fish size) and leaching, means that rates of wastes

accumulation (input minus leaching loss) can be predicted. It is a good start to improve the predictions of waste dispersion models in future work.

Appendix 6.1 The Excel spreadsheet model from part of the 'INPUT DATA' page.



Appendix 6.2 The 'Lookup Table' showing the relationship between fish size, pellet size and settling velocity of feed and faecal pellets.

Pellet Size Lookup				
0.02	1	5.0	4.3	
0.04	2	5.9	5.8	
0.1	3	6.8	7.7	
0.2	4	7.1	9.9	
0.4	6	10.6	12.6	6
1.3	8	10.7	11.5	
2	10	10.1	11.8	
3	12	11.9	12.2	
4	14	12.8	10.9	
Fish Size	Pellet size	Feed A sinking rate	Feed B sinking rate	Faeces sinking rate

Appendix 6.3 Illustrating division of current data (A: raw data by chronological series from current meter, B: the data after sorting by current directions, both are part of spreadsheet)

A

Date	Time	Pressure	Flow	Direction
16/09/97	15:51:24	12.98918	0.151	153
16/09/97	16:11:24	12.8504	0.191	136.4
16/09/97	16:31:24	12.89666	0.188	145.6
16/09/97	16:51:24	13.0817	0.213	132.9
16/09/97	17:11:24	13.20507	0.214	146.2
16/09/97	17:31:24	13.28217	0.208	140.9
16/09/97	17:51:24	13.29759	0.267	136.4
16/09/97	18:11:24	13.32844	0.297	140.9
16/09/97	18:31:24	13.35928	0.295	142.1
16/09/97	18:51:24	13.39012	0.297	131.2
16/09/97	19:11:24	13.39012	0.329	130.3
16/09/97	19:31:24	13.39012	0.338	132.7
16/09/97	19:51:24	13.39012	0.328	131.1
16/09/97	20:11:24	13.3747	0.19	127.9
16/09/97	20:31:24	13.3747	0.113	125.4
16/09/97	20:51:24	13.3747	0.092	132.5
16/09/97	21:11:24	13.3747	0.067	114.8
16/09/97	21:31:24	13.32844	0.073	123.4
16/09/97	21:51:24	13.23591	0.104	334.9
16/09/97	22:11:24	13.29759	0.051	340.6
16/09/97	22:31:24	13.23591	0.095	335.3
16/09/97	22:51:24	13.20507	0.123	336.8
16/09/97	23:11:24	13.17422	0.176	328.3
16/09/97	23:31:24	13.20507	0.122	328.4
16/09/97	23:51:24	13.26675	0.123	335.1
17/09/97	00:11:24	13.28217	0.16	333.2
17/09/97	00:31:24	13.22049	0.143	327.6
17/09/97	00:51:24	13.20507	0.123	334.9
17/09/97	01:11:24	13.26675	0.042	339.3

B

Date	Time	Pressure	Flow	Direction
21/09/97	03:51:24	13.46724	0	1.2
22/09/97	12:11:24	13.62146	0.013	1.2
26/09/97	16:26:58	13.40555	0.013	2.7
18/09/97	13:31:24	13.49808	0.064	4
19/09/97	09:51:24	13.68316	0.054	5.3
04/10/97	11:06:58	13.22049	0.071	5.8
23/09/97	06:11:24	13.52893	0	7.1
18/09/97	01:51:24	13.43639	0	9.4
28/09/97	21:26:58	13.12796	0.058	10.2
04/10/97	13:06:58	13.25133	0.015	10.5
05/10/97	01:06:58	13.22049	0.004	11
20/09/97	02:51:24	13.55977	0	12
01/10/97	00:46:58	13.23591	0	12
26/09/97	03:51:24	13.57519	0.015	12.2
02/10/97	08:46:58	13.28217	0.052	13.1
04/10/97	04:46:58	13.17422	0	13.5
04/10/97	10:46:58	13.20507	0.05	15.9
25/09/97	08:31:24	13.55977	0.004	16.6
30/09/97	12:46:58	13.11254	0	17.6
23/09/97	01:31:24	13.54435	0.01	17.7
18/09/97	02:11:24	13.43639	0	17.8
29/09/97	12:46:58	13.31302	0.006	18.2
27/09/97	16:26:58	13.29759	0	18.8
19/09/97	10:11:24	13.66773	0.058	19.6
17/09/97	01:51:24	13.29759	0.07	20.4
28/09/97	20:06:58	13.14338	0.108	20.7
04/10/97	13:26:58	13.23591	0.026	21.1
24/09/97	03:11:24	13.52893	0	21.3
26/09/97	04:11:24	13.57519	0.022	21.8

Appendix 6.4 Grid data of graphical output of total wastes from part of Excel spreadsheet, where each cell represents 100 m^2 (10 m x 10 m) of cage area. Concentration units are g m^{-2}.

1.5185	0	0	0	0	0	0	0.1147	0	0.2381	0	0.1978	0
0	1.7018	0	0	0	0	0	0.1268	0	0.2654	0	0.216	0
0	0	1.9365	0	0	0	0	0.1417	0	0.3	0	0.2381	0
0	0	0	2.2488	0	0	0	0.1606	0	0.3458	0	0.2654	0.7839
2.6158	0	0	0	2.7271	0	0	0.1853	0	0.4098	0	0.3	0
0	2.9996	0	0	0	4.3435	0	0.219	0	0.5469	0	0.3458	0
0	0	3.5662	0	0	0	15.613	0.2681	0	1.996	0	0.4098	0
0	0	0	5.2945	0	0	0	77.889	0	16.178	0	2.9256	1.1629
0	0	0	0	16.71	0	0	1.8107	627.59	159.65	34.296	1.996	0
0.087	0.0986	0.1138	0.1345	0.1644	79.012	0.2958	16.473	22.638	2391.8	22.455	17.053	0.1686
0	0	0	0	0	0	629.22	159.38	38.383	60.556	1263	159.65	0
0.1842	0.2123	0.2514	0.3099	0.4096	0.649	22.803	2393.4	22.751	19.227	23.01	2501.5	1.0993
0	0	0	0	0	0	4.9638	58.745	1277	160.37	38.383	60.556	0
0.087	0.0986	0.1138	0.1345	0.1644	0.8502	0.2958	2.393	22.638	2574.8	22.455	18.712	0.1686
0	0	0	0	0.4174	0	0	0.9857	4.9638	59.292	1327.7	160.37	0
0.0704	0.0779	0.087	0.4123	0.1138	0.1345	0.1644	1.5543	0.2958	2.841	22.638	2528.8	33.072
0	0	0.2523	0	0	0	0.4174	0.5476	0	1.3648	4.6955	59.292	0
0	0.2114	0	0	0	0.3136	0	0.448	0	1.4798	0	2.3265	0
0.1821	0	0	0	0.2523	0	0	0.3791	0.2683	0.8375	0	1.3648	0
0	0	0	0.2114	0	0	0	0.5202	0	0.7074	0	1.0326	33.404
0	0	0.1821	0	0	0	0.1491	0.2899	0	0.6138	0	0.8375	0
0	0.1601	0	0	0	0.122	0	0.2594	0	0.5428	0	0.7074	0
0.1428	0	0	0	0.1032	0	0	0.2347	0	0.487	0	0.6138	0
0	0	0	0.0894	0	0	0	0.2143	0	0.4419	0	0.5428	0

Appendix 6.5 The predicted production data for Trial 1 (Machrihanish site) taken and appeared from Excel mass balance submodel.

		Date	Fish No.	Size (kg)	SGR	Total Biomass (kg)	Feed Type	Pellet Size (mm)	Feed Rate	Feed rate (%)	FCE
Cage No.	**1**	15/03/97	11250	0.11	0.557	1238	Smolt-30	3	0.01377	1.377	0.404
Cage size	**2250**	15/04/97	10688	0.13	0.894	1389	Smolt-30	3	0.01288	1.288	0.694
(m3)		15/05/97	10634	0.17	0.542	1808	Smolt-30	3	0.01401	1.401	0.387
		15/06/97	10581	0.2	0.744	2116	Smolt-40	4	0.01543	1.543	0.482
		15/07/97	10528	0.25	0.717	2632	Sup-40	4	0.01504	1.504	0.477
		15/08/97	10475	0.31	1.167	3247	Sup-40	4	0.00770	0.77	1.516
		15/09/97	10423	0.44	0.492	4586	Sup-60	6	0.01224	1.224	0.402
		15/10/97	10371	0.51	1.149	5289	Sup-60	6	0.01009	1.009	1.139
		15/11/97	10319	0.72	1.028	7430	Sup-60	6	0.00854	0.854	1.203
		15/12/97	10267	0.98	0.675	10062	Sup-60	6	0.00639	0.639	1.056
		15/01/98	10216	1.2	1.102	12259	Sup-60	6	0.00685	0.685	1.608
		15/02/98	10165	1.67	0.683	16976	Sup-80	8	0.00654	0.654	1.045
		15/03/98	10114	2.05		20734	Sup-100	10	0.00598	0.598	0.000
			1136							**1.04**	
			Mortality		**0.81**					**mean annual % feed intake**	
					mean annual SGR						
							FCE =	**0.78**		**1.28**	
										mean annual FCR	

Appendix 6.6 Mass balance of predicted carbon loadings taken from Excel spreadsheet for Trial 1, Machrihanish production data.

MASS BALANCE CALCULATIONS	Based on annual production data of data	Assumed 1 ton of annual production
Annual production (kg):	19497	1000
FCR:	1.28	1.28
Waste food (% as dec.)	0.05	0.05
C in food (% as dec. of 8% moisture)	0.46	0.46
C in flesh (% as dec.of 73% moisture)	0.143	0.143
C respired (% as dec.)	0.5	0.5
Carbon input (kg/yr)	11449	589
Carbon in waste feed (kg/yr)	572	29
Carbon consumed (kg/yr)	10877	559
Carbon harvested (kg/yr)	2788	143
Carbon respired (kg/yr)	5438	280
Total Carbon wasted(consume-harvest-respire+waste feed)	3223	166

Appendix 6.7 Mass balance nitrogen loadings of Excel spreadsheet for Trial 1, Machrihanish production data.

MASS BALANCE CALCULATIONS	Based on annual production of data	Assumed 1 ton of annual production
Annual production (kg):	19497	1000
FCR:	1.28	1.28
Waste food (% as dec.)	0.05	0.06
N in food (% as dec. of 8% moisture)	0.074	0.074
N in flesh (% as dec.of 73% moisture)	0.028	0.028
Other output(% as dec. of sedimentation, NH3 excretion etc.)	0.44	0.44
Nitrogen input (kg/yr)	1847	95
Nitrogen in waste feed (kg/yr)	92	5
Nitrogen consumed by fish (kg/yr)	1734	90
Nitrogen harvested (kg/yr)	546	28
Other output (kg/yr)	743	39
Total Nitrogen wasted (consume-harvest-other output +waste feed)	558	28

Appendix 6.8 The predicted production data for Trial 2 (Aquascot cage site) taken from Excel mass balance submodel.

		Date	Fish No.	Size (kg)	SGR	Total Biomass (kg)	Feed Type	Pellet Size (mm)	Feed Rate	Feed rate (%)	FCE
Cage No.	**1**	31/1/97	14871	0.0646		961	Smolt-30	2	0.006	0.6	
Cage size	**2250**	28/2/97	13863	0.0937	1.240	1299	Smolt-30	2	0.012	1.2	2.066
(m3)		31/3/97	13795	0.1076	0.461	1484	Smolt-30	3	0.014	1.4	0.384
		30/4/97	13698	0.1605	1.333	2199	Smolt-40	3	0.014	1.4	0.952
		31/5/97	13662	0.2287	1.180	3124	Sup-40	4	0.016	1.6	0.843
		30/6/97	13646	0.3264	1.186	4454	Sup-40	4	0.016	1.6	0.741
		31/7/97	13632	0.5208	1.557	7100	Sup-60	6	0.016	1.6	0.973
		31/8/97	13584	0.8315	1.560	11295	Sup-60	6	0.012	1.2	0.975
		30/9/97	13516	1.3141	1.526	17761	Sup-60	8	0.011	1.1	1.271
		31/10/97	13448	1.8124	1.072	24374	Sup-60	8	0.014	1.4	0.974
		30/11/97	13381	1.9552	0.253	26163	Sup-60	8	0.009	0.9	0.181
		31/12/97	13314	2.5536	0.890	34000	Sup-80	10	0.009	0.9	0.989
		31/1/98	13248	3.1496	0.699	41725	Sup-100	12	0.007	0.7	0.777
		Mortality	**1623**							**1.2**	
					1.080					**mean annual % feed intake**	
					mean annual SGR						
						FCE =	**0.899726**	**1.11**			
								mean annual FCR			

Appendix 6.9 Mass balance portion of Excel Spreadsheet for Loch Duich cage site.

MASS BALANCE CALCULATIONS					
Total food input (g)	11935276				
FCR:	0.98				
Waste food (%)	10				
Carbon component in food (%)	46				
Carbon component in flesh (%)	14.3				
Carbon component respired (%)	50				
Carbon input	5370260				
Carbon in waste feed	537026				
Carbon consumed	4833234				
Carbon harvested	1706745				
Carbon respired	2416617				
Carbon in faeces	709872				
			SETTLING RATES		
Number of cages:	5		Food (m/s)	0.106	
Depth below cages:	12		Faeces (m/s)	0.05	
FOOD INPUTS					
Feeding rate (% B.W. per day)	1.042		Feeding period	10	days
Cage identifier:	Fish biomass (kg)	Food (g)	factor		
1	32625	3399525	0.219429119		
2	13665	1423893	0.52388401		
3	23812	2481210.4	0.300641483		
4	20268	2111925.6	0.353210726		
5	24172	2518722.4	0.29616395		
6					
7					
8					
9					
10					
11					
12					
13					
14					
15					
16					

Chapter 7

General Discussion

7.1 Introduction

The objectives of the present study have been met in that the physical characteristics of a range of feed and faecal pellets, typical of the current aquaculture scene in Western Europe, were determined and incorporated into the wastes dispersion model that have been developed at the IoA and Dunstaffnage Marine Laboratory (C. Cromey, pers. comm). Nutrient leaching rates of faecal material were also taken into account in the development of the IoA model. All the empirical data including the saltation measurements of feed pellets derived from this thesis would be helpful for other modellers to incorporate into their models, *e.g.* BenOss dispersion model (Cromey *et al.*, 1997) and DEPOMOD model (Cromey *et al.*, 2000). It remains of paramount importance to distinguish between uneaten food and faeces because both their physical characteristics and settling velocities are different, determining the horizontal dispersion and the amounts reaching the sea bed. With the verified new model incorporating parameters determined in this thesis, the dispersion area can be more accurately predicted.

While there are definable relationships between pellet size and settling velocity, these are inadequate for the purposes of generating algorithms. It is thus proposed that models employ a 'look-up-table' approach to differentiating between diets under various environmental conditions. In addition to collecting a comprehensive set of data for a range of feeds and environmental conditions, further investigations into changes in the physical environment with depth and the fate of waste feed pellets within salmon cages are necessary.

All of the models discussed previously require first of all that there are sufficient data to estimate loadings. These can be estimated from minimal data, even from production figures alone, although the more data on stock composition and feeding the better as it provides more accurate estimates of the waste outputs that can then be incorporated into the dispersion model. Models of benthic deposition require both bathymetric data, which is usually available, and current data. Detailed current information such as required by Gowen *et al.* (1994) is seldom readily available, unless a hydrodynamic survey at the site has been carried out. Until now modellers have incorporated the mean current values over the period of observation, as is the case in the model of Silvert (1994). Although it is likely to have more accurate prediction of benthic impacts if the individual current value was incorporated into the model, the cost and the time spend in the modification of the model would be anticipated very high.

7.2 FCR and uneaten food

The composition and physical nature of aquaculture wastes reflects the composition of the diet and digestibility of its components. The first challenge of managing aquaculture waste clearly lies in nutritional strategies of 'reduction at the source', namely improving the digestibility of feeds and balancing nutrient and energy requirement (Cho, 1998). Also improvements in feed conversion and reduction in waste losses to the environment can be achieved by a better understanding of feeding behaviour and optimization of feeding regimes. All of these have been addressed since the mid-1980s, the development of high energy diets has led to most commercially available feeds being very energy-dense and having a high coefficient of digestibility. Hence, today's principal challenge is not to

overestimate feed requirements, but improve the feed efficiencies by good husbandry and management. In the last 20 years, the FCRs for salmon aquaculture systems in Europe have been reduced by about 50%, resulting in an 80% reduction in waste discharges (Lopez-Alvarado, 1997). In the present study, although the FCR is slightly higher in the small-scale experimental system (Machrihanish) than at the cage farms (Aquascot Ltd. and Loch Duich Marine Harvest Ltd.), especially for larger fish, it is unlikely that this is because there was more uneaten food in the former. It is more probable that there were some underlying physiological explanations such as fish under cage farming may have less stress than those under land-based culture. Nevertheless, commercial sites often overestimate growth in order to make their FCRs seem better than they really are. Assuming the average FCR value for Scotland is currently 1.2 and that the proportion of uneaten food is 10%, with 100,000 t of annual production, about 120,000 t of feed are used. Some 108,000 t is thus consumed, and 12,000 t feed input may be attributed to the wasted food. If a figure of 300 g per kg of food ingested is assumed as faecal wastes (Beveridge *et al.*, 1991), then 32,400 t (108,000 x 30%) of faecal waste is produced. In sum, 44,400 t of solid wastes are estimated as being deposited into the Scottish coastal environment. If an FCR of 1.0 is assumed and with the same uneaten food proportion of 10% being wasted, faecal wastes will be reduced due to improvements in food digestibility, *e.g.* we may suppose 20 % of fed amount ends up as faecal waste (*c.f.* Cho and Bureau, 1997), thus the waste loadings from uneaten food and faeces will result in only 30,000 t of solid wastes. If uneaten food proportion reduced to 5%, then only 25,000 t of solid wastes are produced. Unfortunately, the relationships between FCR, uneaten food losses, digestibility and faecal production remain poorly explained. Nevertheless, the above examples and three case studies stated in Section 6.5 (Table 6.2)

illustrate that small changes in FCR can have a significant effects on waste loadings, and underline the importance of improving feed efficiency and feeding methods. In practice, the uneaten food can be reduced to a minimum if husbandry practices are carefully monitored. It might, in this context, be useful to employ video acoustic equipment to monitor feeding behaviour and fish satiation, thereby also determining the extent of feed wastage (Juell, 1991; Foster *et al.*, 1995).

In mariculture, the majority of the uneaten food is discharged from the farm and sinks rapidly towards the sea floor. Some may be consumed by wild fish. These are difficult to observe and quantification of consumption is difficult (*e.g.* Smith *et al.*, 1997; Dougall and Black, 1999). Hence, there is likely to be an overestimation of the waste output from the cage farm that impacts on the sea bed and it would be a significant improvement if the mass-balance model could be modified to take account of this important factor.

7.3 Nutrient loss to the environment after sedimentation

Using a mean settling velocity of 10 cm s^{-1} (see Section 3.3), food pellets take approximately 5 min to settle out of a 30 m water column typical of the majority of Scottish cage sites. It can thus be assumed that, given the tidal currents that prevail at most Scottish sites (3-5 cm s^{-1}), there is likely to be little horizontal displacement of uneaten food pellets. The rapid settling velocities and the associated high water stability mitigate against any significant loss of carbon and nitrogen into solution as pellets settle out of the water column. It also seems unlikely that within that time scale there would be any significant loss of carbon and nitrogen from pellets as a result of microbial activity (D. McLusky pers.

comm.). Thus uneaten food pellets probably arrive at the sea-bed with their original composition intact, loss of soluble components by leaching being negligible. To test this assumption, food pellets were soaked in sea water (28 psu, 10 (± 2) °C) for 72 h, the proportions of carbon and nitrogen leached from feed pellets after 3 d immersion were less than 3% and 19%, respectively (Chen *et al.*, unpub. data). However, the leaching experiments of faecal pellets demonstrate that nutrient leaching from faeces is rapid, posing problems for waste management strategies that focus on the recovery of faecal wastes to reduce environmental impact. One of the assumptions in the present models is that the nutrient content of faeces does not change as they settle. Models for dispersion of solid material that take no account of losses by leaching will overestimate nutrient inputs to sediments. Fortunately, we are now able to deduct the leaching proportion from the mass balance equation to have more accurate predictions of waste output. In addition, there may be an important input of dissolved nutrients to the water column that require further research, *e.g.* in relation to algal blooms.

The periods of three and seven days deployment for sediment traps without preservation chemicals may have been slightly too long (D. Angel pers. comm.), and sedimentation measurements may therefore have been affected by artifacts such as seston, microalgae, decomposition and fouling. Mass transport of wastes by benthic infauna is probably not significant in relation to other transport mechanisms because of the small amount of biomass in organisms with respect to their surroundings. However, in the surface of sediments, especially near cages, bioturbation is a very significant transfer mechanism (Pearson and Rosenberg, 1978; Weston, 1990; Johannessen *et al.*, 1994; Dougall and Black,

1999) which might be appropriately included in consideration of the mass balance formulation. The values obtained could have been compared with values from control sedimentation traps deployed some distance, for example 100 m, from the cages. It may also be more appropriate to employ traps for only one or two days to reduce the uncertainty in future studies.

7.4 Carrying capacity and nutrient loadings

Since most marine cage farms in Europe use dry pellets as a food source, a general mass balance equation can be used to calculate the nutrient load and waste dispersion area if details of the production cycles, fish sizes, temperature, water currents, different element content in feed, and different methods of production are known. Nutrient enrichment plays a major role in determining carrying capacity, depending on existing background levels and whether primary production is nutrient limited. This is extremely important in subtropical or tropical farming areas as those enrichment nutrient may help promote algal blooms, which can result in disastrous damage to fish farms and deterioration of the immediate environment. However, it is difficult both to calculate and to set reasonable criteria for allowable changes in nutrient levels. Hargrave (1994) suggested an assimilative capacity for sediments based on a threshold sedimentation rate of 1 g C m^{-2} d^{-1} in temperate areas, estimated from his field work in Canada. However, it does not necessarily apply to the present study site due to site-specific differences in environmental conditions such as temperature, bathymetric and current data. Nevertheless, if comparing the indicative figure of Hargrave (1994) with the present study at the Loch Duich site and assuming the dispersal areas given in the present model (*i.e.* area 200 m radius which accounts for 125,663 m^2), the

total carbon loadings from five cages over a 10 d period is 1,342,379 g C, resulting in an average loading of 1.02 g C m^{-2} d^{-1}, which closely approximates the assimilative value given by Hargrave (1994). No adverse impact has been reported from our study site with the present production figure. If Hargrave's figures are valid, then the stocking density at the site of 10.18 kg m^{-3} is close to maximum value, resulting in a total carrying capacity 23 tonnes per cage (15 m x 15 m x 10 m). However, to ensure the sustainability of the cage site, it would be useful to check if microbial mats are present in the sediments and if so, how far the abiotic zone extends from the farm.

7.5 Implications for Taiwanese cage aquaculture

Since impacts from intensive cage farming have mainly been studied in temperate areas, there is a need to increase the understanding of environmental effects from intensive cage farming in the tropics (Beveridge and Phillips, 1993; Beveridge, 1996; Wu *et al.*, 1994, 1999). The few studies conducted in the tropics (*e.g.* Angel *et al.*, 1992; Choo, 1994; Troell and Berg, 1997, Wu *et al.*, 1999) have indicated less negative effects from fish cages on the local environment compared to temperate regions. Even if this indicates a greater capacity for tropical systems to assimilate aquaculture wastes, the data are at best insufficient and inconclusive. Moreover, severe impacts from fish cage farming have also been documented in the tropics (Wu, 1995). The success of aquaculture in Taiwan over the years has resulted in tremendous capital investment and the development of advanced techniques and technologies. However, after the collapse of the shrimp farming industry in the late 1980s, the government has devoted much effort to the development of modern offshore fish

farming as a top priority for its national aquaculture plan. The recent and continuing expansion of cobia (*Rachycentron canadum*), grouper (*Epinephelus* spp.) and sea bream (*Pagrus major*) cage farming in inshore areas, and the intention to develop offshore potential, has led to questions on the degree of the ecological impact that can be anticipated in the future (Chen, 2000). The high nitrogen and carbon concentrations of cage aquaculture discharges are a potential threat to the environment in the vicinity of cage farms. However, in the absence of legislative control, there is little economic pressure to treat wastes and comparatively few farms have any form of waste treatment in Taiwan, especially cage farms which normally discharge wastes directly into the immediate environment. To ensure the sustainability of the new industry, it is essential to develop methods for predicting the effects of fish farms on their surroundings. Although offshore cage farming in its initial stage is unlikely to raise environmental concerns, an example using the revised waste dispersion model to predict the waste loading and estimate the carrying capacity or stocking density of the site would be very helpful. Being able to predict aquaculture wastes from a fish farm is beneficial to both the farmer and the government regulator. Farmers can predict the feed used to grow his fish, while at the same time, government or regulators can use the model to predict environmental enrichment resulting from the operation of farms in their jurisdiction. Careful site selection can ensure minimum impact on sensitive areas; visually, in terms of the ecosystem, individual species, and the increasingly popular eco-tourism industry. Hopefully this work on the waste dispersion modeling has some implications for the Taiwanese cage industry.

7.6 The future-predicting the environment

Although there are several empirical and mechanistic models available for predicting the input of organic matter from marine cage farms to the sea bed, quantitative connections between input and ecological changes have not yet been developed (GESAMP, 1996). At present environmental models have not been developed to the point where they can be used as the only tool to predict impacts in the environment. They must complement field monitoring. It is important to understand the limitations of a model as output results may have different interpretations depending on site characteristics. Environmental concerns of cage aquaculture are in line with the key changes in scientific analysis of the behaviour of benthic fauna in the vicinity of cage sites. There has been a shift from description of changes (*e.g.* loss of biodiversity, changes in benthic community structure) to predictive models based on an understanding of underlying processes. Our understanding of the interactions between fish farms and the environment has reached the stage where it is reasonable to expect that quantitative estimates of the possible environmental consequences of aquaculture development can be provided for regulatory and mitigation purposes. It is essential that the continuing usefulness of this model is maintained and enhanced. It is my hope that this study and program will prove to be useful to all who use it.

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