



REVIEW

Discharge of nutrient wastes from salmon farms: environmental effects, and potential for integrated multi-trophic aquaculture

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ABSTRACT: We quantified release rates of carbon (C), nitrogen (N) and phosphorus (P) waste from Norwegian salmon farms in 2009 in order to evaluate the theoretical environmental influence on surrounding waters and the potential for integrated multi-trophic aquaculture (IMTA) driven by salmon aquaculture. Of the total feed input, 70% C, 62% N and 70% P were released into the environment, equivalent to an annual discharge of about 404 000, 50 600 and 9400 t of C, N and P, respectively, based on total salmon production of 1.02×10^6 t. We predicted that 48% of feed C was respired as CO₂, 45% of feed N was excreted as dissolved inorganic N (DIN), and 18% of feed P was excreted as dissolved inorganic P (DIP). Approximately 44% of feed P was released as particles, dominating solid wastes. The mean food conversion ratio (feed supplied per fish produced) of Norwegian salmon farms was 1.16 ± 0.08 SE in 2009. Estimates of the potential for IMTA driven by salmon farming showed a far higher potential for seaweed production based on the released DIN than for mussel production based on released appropriately sized particulate organic carbon (POC). The daily volumetric loading rates of DIN from salmon farms (range for counties: 40 to 501 $\mu\text{g N m}^{-3} \text{d}^{-1}$) were <15% of the natural loading rate of nitrate from deep water, suggesting that the nutrient loading rate is within safe limits.

KEY WORDS: Cage aquaculture · Atlantic salmon · Nutrient wastes · Feed conversion ratio · FCR · Integrated multi-trophic aquaculture · IMTA · Seaweed · Blue mussels

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INTRODUCTION

Global aquaculture is developing rapidly and is the fastest growing food producing sector (Troell et al. 2009, Abreu et al. 2011), with nearly half of the world's seafood supply now sourced from aquaculture (FAO 2010). Cage aquaculture has been practiced for years in many countries worldwide, and technology has developed steadily over the last few

decades (Enell 1995, Islam 2005). For example, global aquaculture production of marine Atlantic salmon *Salmo salar* and rainbow trout *Oncorhynchus mykiss* increased from 0.92×10^6 t in 1999 to 1.74×10^6 t in 2009 (FAO, online query: www.fao.org/fishery/statistics/global-aquaculture-production/query/en).

In Norway, intensive open cage culture with a continuous water exchange with surrounding waters

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(Troell & Norberg 1998) has expanded significantly over the last few decades (Skogen et al. 2009). Cage aquaculture can release a considerable amount of biogenic waste such as organic wastes and inorganic nutrients that are generated in the production process (Troell & Norberg 1998, Cheshuk et al. 2003, Vassallo et al. 2006, Redmond et al. 2010). The rapid expansion of cage aquaculture has raised a general concern about increasing amounts of solid and dissolved nutrients released to the aquatic environment (Perez 2002, Whitmarsh et al. 2006, Redmond et al. 2010, Azevedo et al. 2011, Skriptsova & Miroshnikova 2011).

Fish farming releases carbon (C), nitrogen (N) and phosphorus (P) waste (Fig. 1). Dissolved inorganic N (i.e. NH_3^+) and P (i.e. PO_4^{3-}) (DIN and DIP, respectively) are released through excretion, and inorganic C as CO_2 is released through respiration. Particulate organic C, N and P (POC, PON and POP, respectively) are released through defecation and loss of feed. Dissolved organic C, N and P (DOC, DON and DOP, respectively) are generated through dissolution

of particulate organic fractions (Olsen & Olsen 2008).

These different nutrient components have the potential to influence different parts of the marine ecosystem. Inorganic nutrients such as DIN and DIP are readily available for phytoplankton and macroalgae (Troell et al. 2003, 2009) and uptake can be fast, although it may take several days before the phytoplankton biomass responds to this increase (Olsen & Olsen 2008). Large faecal particles and uneaten feed sink rapidly and may accumulate in sediments on the seafloor (Cromeey et al. 2002, Olsen & Olsen 2008, Nickell et al. 2009) where they may be consumed by detritus-eating animals. Small particles of waste can remain in suspension and then be consumed by filter-feeding zooplankton or by visual feeders, such as fish, in the water column, or by mussels (Olsen & Olsen 2008, Troell et al. 2009). DON and DOP are dissolved from feed and faecal particles, and may constitute small N- and P-containing molecular species such as, for example, amino acids and nucleotides, but mostly comprise complex dissolved chemical compounds ($<0.2 \mu\text{m}$) that are avail-

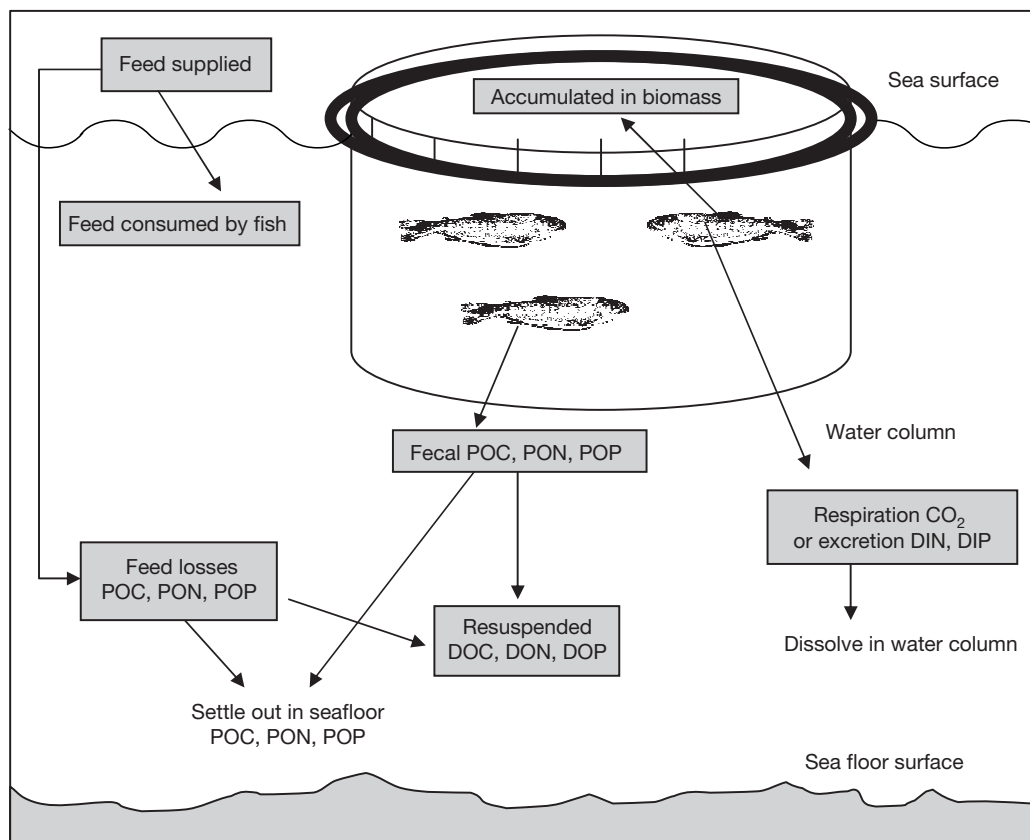


Fig. 1. The flow and fate of nutrient components from a salmon cage system. Dissolved inorganic nitrogen and phosphorus (DIN and DIP, respectively) are released through excretion, and inorganic carbon (CO_2) through respiration. Particulate organic C, N and P (POC, PON and POP, respectively) are released through defecation and feed loss. Dissolved organic C, N and P (DOC, DON and DOP, respectively) are resuspended from faeces and feed particles

able for bacteria and for phytoplankton over a longer time scale (Palenik & Morel 1990, Fan et al. 2003, Stoecker & Gustafson 2003).

One of the main challenges facing aquaculture today is sustaining a continued increase in fish production while minimizing the impact on the environment (Sugiura et al. 2006, Navarrete-Mier et al. 2010). The salmon aquaculture industry has taken a number of steps to reduce nutrient release from salmon farming facilities. These efforts include optimizing feed composition and improvements in feed digestibility and feeding technology (Cheshuk et al. 2003, Islam 2005). These measures reduced nutrient loading and mitigated pressure on the environment. These improvements of environmental technologies of cage culture have been significant in European aquaculture over recent decades; for instance, the mean economic feed conversion ratio (FCR) for the Norwegian salmon industry was 2.08 in 1974 but can now reach as low as 1.0 to 1.1 (Enell 1995, Piedrahita 2003, Islam 2005).

Integrated multi-trophic aquaculture (IMTA) is a practical and viable solution for mitigating the possible negative environmental impacts of waste produced by fish aquaculture. It works by exploiting fish waste as a food resource for extractive and filter feeding species at lower trophic levels, thereby also giving an added value to the investment in feed for cage aquaculture (Barrington et al. 2001). IMTA has been practiced for centuries in Asia (Li 1987, Fang et al. 1996, Qian et al. 1996), where it is now commercially successful at industrial scales. An example is the cultivation of scallop, kelp and abalone in the marine IMTA system of Sungo Bay, China, (Fang et al. 1996, Troell et al. 2009). The approach is now also becoming widely accepted in western countries (Troell et al. 2009, Abreu et al. 2011, MacDonald et al. 2011) and several pilot experiments using IMTA have recently been conducted in Canada, Scotland and Australia (Stirling & Okumus 1995, Cheshuk et al. 2003, Barrington et al. 2010). In the Bay of Fundy, Canada, blue mussels *Mytilus edulis* and kelps (*Saccharina latissima* and *Alaroea esculenta*) reared close to Atlantic salmon cages exhibited growth rates that were 46 and 50% higher, respectively, than at control sites. The products are now being sold commercially, adding value to salmon production (Reid et al. 2009, Troell et al. 2009).

Norway has more recently made efforts to develop IMTA, driven by waste generated by the large salmon producing sector (e.g. Handå 2012). It is important to quantify the different nutrient components released by fish farms in order to fully understand the

potential of IMTA and the possible effects on the coastal ecosystems, and to optimize the design and nutrient recovery efficiency of IMTA systems (Reid et al. 2009, Dalsgaard & Pedersen 2011). The objectives of the present study were to quantify the release rate of inorganic and organic nutrient wastes generated in Norwegian salmon aquaculture, and to make a preliminary evaluation of nutrient loading rates of coastal waters and the theoretical potential of salmon-driven IMTA.

MATERIALS AND METHODS

Estimating release rate of wastes

The release rate of waste from cage aquaculture can be estimated using a simple mass balance principle, provided that the required input information is available (Sturner & George 2000, Olsen & Olsen 2008, Reid et al. 2009). The main data input used to estimate the release rates of particulate and dissolved C, N and P wastes were monthly statistics on feed used and fish production made available by the Norwegian Fish Farmer Association (FHL). Further required inputs were: data on the digestibility (assimilation efficiency) of C, N and P components of the feed and feed loss, obtained from feed producers and the published results of other studies; data on C, N and P stoichiometry of fish and feed that were derived from other studies and our own measurements; and data on water content of feed and fish taken from published literature. The values of the coefficients used in the mass balance calculations are listed in Table 1.

Mass balance of C, N and P in fish

The mass balance for fish (f) consuming food can be represented by:

$$I_f = A_f + F_f = G_f + E_f + F_f \quad (1)$$

where I is food intake, A is assimilated food, F is defecation, G is growth or retention in biomass and E is excretion. The assimilated food is the part that is digested by fish and taken up in tissues, and assimilation per ingestion is termed assimilation efficiency (AE) and is compatible to the digestion efficiency reported by feed manufacturers. The AE for C, N, or P was calculated as:

$$AE_f = A_f / I_f \quad (2)$$

Table 1. Values of model coefficients for water content of feed and fish, the assimilation efficiency (AE) of feed carbon (C), nitrogen (N) and phosphorus (P), the content of C, N and P in feed and fish, and the leaching rate (i.e. soluble fraction) of faecal and feed C, N and P. DW: dry weight; WW: wet weight

Coefficient	Value	References
Feed loss (%)	3	Corner et al. (2006), Reid et al. (2009)
Water in feed (%DW)	4	Piedecausa et al. (2010)
Dry matter in fish (%WW)	33.5	Talbot et al. (1986)
Carbon (C)		
AE	0.8	Cheshuk et al. (2003), Corner et al. (2006)
C content in feed (%DW)	51	Petersen et al. (2005), Corner et al. (2006)
C content in fish (%DW)	50	Olsen & Olsen (2008)
Soluble fraction (%DW)	15	Chen et al. (2003)
Nitrogen (N)		
AE	0.85	T. Lea, Skretting AS pers. comm.
N content in feed (%DW)	7.2	Gillibrand et al. (2002), Mente et al. (2006)
N content in fish (%WW)	3	Ackefors & Enell (1990), Davies & Slaski (2003)
Soluble fraction (%DW)	15	Chen et al. (2003)
Phosphorus (P)		
AE	0.5	Reid et al. (2009), Bureau et al. (2003)
P content in feed (%DW)	1.2	Petersen et al. (2005), Reid et al. (2009)
P content in fish (%WW)	0.40	Talbot et al. (1986)
Soluble fraction (%DW)	15	Sugiura et al. (2006)

for each individual element. The C, N and P assimilated in the food can be used for growth and weight increments or may be released through respiration and excretion. The growth efficiency (GE) is an ecological term and expresses production yield, and is calculated as:

$$GE_f = G_f / I_f \quad (3)$$

for each element with production and consumption expressed in the same units (e.g. $g\ d^{-1}$). Excreted C is CO_2 released in respiration, whereas N and P are mainly excreted in the form of ammonia (NH_3) released over the gills and PO_4^{3-} in urine. The amount of nutrients excreted (E_f) is equal to the amount of assimilated nutrients minus the amount of nutrients retained in fish biomass, and can be represented by the following equation:

$$E_f = A_f - G_f = (I_f \times AE_f) - G_f \quad (4)$$

for each individual element, where I is the total dry feed intake multiplied by feed content per dry weight (DW) of the nutrient concerned (C, N or P). AE for C,

N and P can be obtained from feed manufactures and literature (Table 1). G is the C, N and P harvested as fish biomass and can be calculated as fish production multiplied by C, N and P contents in fish. Of the food ingested, a proportion will not be digested by fish, but released as faecal particles. The indigestible content in the feed will mainly determine the fraction of faeces produced per feed consumed. F_f can be estimated as:

$$F_f = I_f - A_f = I_f \times (1 - AE_f) \quad (5)$$

Mass balance of C, N and P on the fish farm level

The waste emission from fish farms differs from that from individual fish because a proportion of feed will be lost as particles and some fish individuals will die (Davies & Slaski 2003, Olsen & Olsen 2008). Dead salmon are usually collected for destruction and land deposition, so are not lost directly to the environment. In our calculations, mortality is a part of fish production, and we have therefore assumed that the additional mortality is insignificant in the context of the overall transfer of matter.

Feed losses previously represented a significant source of particulate waste from fish farms (20 to 40 % of total feed input; Beveridge 1987, Islam 2005, Reid et al. 2009), but feeding is now better controlled and losses are lower than in the past (Cromey et al. 2002, Bureau et al. 2003, Davies & Slaski 2003, Mente et al. 2006). In our calculations, we assumed that 3 % of the total feed input was lost uneaten. According to this, the feed input to the farm (I_F , 'F' for farm level) equals total fish assimilation (A_F) and defecation (F_F) plus the feed loss (L_F):

$$I_F = A_F + F_F + L_F \quad (6)$$

The loss of particulate organic material of element X (L_{POX}), where X is C, N or P, from the fish farm equals the total defecation (F_F) plus the feed loss (L_F) of the element:

$$L_{POX} = F_X + L_X \quad (7)$$

A fraction of the particles will dissolve and become dissolved organic material (DOX). According to Chen et al. (2003), approximately 15 % of the C and N in faeces were leached after a few minutes in water, and there was no further significant leaching thereafter. Moreover, approximately 15 % of faecal P becomes soluble in minutes or hours (Sugiura et al. 2006). Feed loss is relatively low from modern salmon farming where feeding is normally camera assisted

and losses are monitored using either camera or acoustic devices installed to detect uneaten pellets. We assumed the dissolution rate of lost feed particles to be the same as that for faeces in our mass balance model. The loss of dissolved organic material (L_{DOX}) can accordingly be calculated as:

$$L_{DOX} = L_{POX} \times SF \quad (8)$$

where SF is the soluble fraction per dry weight of faeces and feed particles. The net loss of particulate matter (L_{NPOX}) then equals particulate waste nutrients (L_{POX}) minus dissolved organic nutrients (L_{DOX}) leaching from particulate matters generated through defecation and feed loss, i.e.:

$$L_{NPOX} = L_{POX} - L_{DOX} \quad (9)$$

Values of coefficients

The input values used in the calculations are reviewed in Table 1. Salmon feed pellets have been reported to have a water content of 4 % of DW (Piedecausa et al. 2010). DW per wet weight (WW) of Atlantic salmon has been determined in the literature to be 34 % (Talbot et al. 1986). The proportion of feed that is lost varies with type of feed, feeding regimes and regions. Early studies in open cage aquaculture revealed loss rates in the range of 1 to 40 % for different feed types and feeding regimes (Gowen et al. 1985, Beveridge 1987, Hall 1990, Troell & Norberg 1998, Islam 2005). In a more recent study of feeding assisted by modern technology, feed loss was found to be below 5 % of the input (Cromey et al. 2002, Bureau et al. 2003, Reid et al. 2009). The Norwegian salmon farmers claim that feed loss in modern salmon production, using camera assisted feeding control and acoustic registration of lost feed pellets, is negligible and that there are no economic and environmental incentives to further reduce feed loss. Therefore, a feed loss rate of 3 % appears to be a reasonable assumption for modern salmon production (Cromey et al. 2002, Corner et al. 2006, Reid et al. 2009). The average whole body P composition in salmonids was reported to be 0.4 % of WW (Talbot et al. 1986). The average values for digestibility (i.e. AE) of feed ingredients, the C, N and P composition of feed and fish, and the soluble fraction of C, N and P from resuspended particles were taken from the literature, fish farm managers and from feed producers (Table 1).

The accuracy of the input data can be important for model performance. Tests of sensitivity for specific

input variables, such as digestibility of N and P components in the feed and feed loss rate, were undertaken to elucidate the dependency of model predictions of dissolved inorganic and particulate wastes on these input variables.

Potential of macroalgae and blue mussel production

Macroalgae have the ability to take up and remove dissolved inorganic nutrients from fish farms in open-water IMTA systems (Buschmann et al. 1996, Martínez-Aragón et al. 2002, Kraemer et al. 2004). DIN ($\text{NH}_4^+ + \text{NO}_3^- + \text{NO}_2^-$) is the limiting factor determining the productivity of seaweed in many marine environments (Troell et al. 2009, á Norði et al. 2011). The removal rate of DIN released from fish tanks varies from 35 to 100 % at different densities and under different experimental designs (Buschmann et al. 1996, Troell et al. 2003). However, the removal rates of macroalgae in an open-water IMTA system might be lower due to the high dilution. Troell et al. (1997) reported that 1 ha of *Gracilaria chilensis* co-cultured in a salmon farm with an annual production of 227 t, and a stocking density of 0.5 kg WW m^{-2} at 2 depths (1 and 3 m), could remove approximately 6.5 % of the dissolved nitrogen from the fish farm. Higher cultivation densities (1.5 to 7 kg WW m^{-2}) of macroalgae were used to remove dissolved inorganic nutrients in fish tank effluents in previous studies (Buschmann et al. 1994, Abreu et al. 2011). To evaluate the potential for seaweed production co-cultured with salmon in open-water IMTA systems, we assumed that 10 to 30 % of total DIN released from salmon aquaculture can be assimilated by macroalgae. Production of seaweed co-cultured in open-water IMTA systems was assumed to be 70 t fresh seaweed ha^{-1} (Yang et al. 2006).

Previous studies demonstrated that suspension feeders such as blue mussels are capable of consuming particles of appropriate size released from fish farms (Stirling & Okumus 1995, Peharda et al. 2007, MacDonald et al. 2011). Wong & Piedrahita (2000) demonstrated that 30 % of the total particulate waste released from rainbow trout (20 to 40 cm length) is suspended solids. Blue mussels can filter particles of sizes between 2 and 1000 μm with high retention efficacy when feeding on particles $\geq 4 \mu\text{m}$ (Møhlenberg & Riisgård 1978, Reid et al. 2009). Elberizon & Kelly (1998) reported that around 70 % of the total solid waste (uneaten feed and faeces) from salmon (mean salmon weight 25 g) held in tanks passes through the

1000 μm meshes. For blue mussels, we assumed that only 10 to 30% of the suspended solids (30% of the total particulate) were of an appropriate size for mussels to capture and consume in an open-water IMTA system. We further assumed a mussel growth yield of 20% for the consumed suspended solids (Olsen et al. 2011). These assumptions together imply that 0.6 to 1.8% of the total particulate waste from salmon farming can be incorporated into mussel biomass.

The mean N content of macroalgae was assumed to be 4.2% of DW (Yang et al. 2006), and the C content of dry flesh of blue mussel was assumed to be 37% (Smaal & Vonck 1997). The dry matter contents of fresh seaweed and blue mussel from our own experiments were approximately 15% and 25%, respectively.

Volumetric loading rate of inorganic nutrients

The volume of coastal water into which the inorganic nutrients from salmon farms are drained cannot easily be estimated. Future management will require an ecosystem approach, but there are few natural physical borders along the Norwegian coast that can serve as boundaries to demarcate separate coastal ecosystems of a size suitable for the kind of ecosystem-based management approaches implemented elsewhere in Europe (Tett 2008). As a first approximation, we have assumed that the nutrients released from salmon farming within the 9 counties that cover the entire west-facing coast of Norway (Norwegian Sea/North Sea) where salmon production is undertaken, are drained into national coastal waters of each county. We calculated the area of these coastal waters as sea area inside Norwegian territorial waters (available from Statens Kartverk, www.norgeskart.no) minus the area between territorial waters and the baseline, which was estimated from maps using image analysis. The volume of the surface mixed layer (to 15 m depth) of these receiving waters was estimated as area \times depth. These are the euphotic waters where photosynthesis takes place and into which most nutrients drain. The volumetric loading rates of inorganic N and P in euphotic waters originating from salmon farming in the counties were finally estimated as mass of inorganic nutrients released per time and volume of receiving waters.

Statistics

Temperature, growth rate, FCR and GE were tested for statistical significance using a 1-way

analysis of variance (ANOVA) with Tukey's multiple comparison tests, and differences were considered significant at the $p < 0.05$ level. Means are given with standard error. The statistical analyses were performed by SPSS (17.0) for Windows. Tables were made using Excel 2003, and figures using Sigmaplot 10.0.

RESULTS

Salmon production and feed conversion

The total amount of feed pellets used in all Norwegian salmon farms was 1.17×10^6 t, and total salmon production was 1.02×10^6 t WW in 2009, according to FHL statistics. Generally, the sea temperature of mixed euphotic waters (0 to 15 m depth) for all counties began to increase in April, reached its highest value in August, and then gradually decreased from September (Fig. 2A). The average annual temperature over the year decreased from south to north (Fig. 2B).

Fig. 2C shows the amounts of seasonal feed used (DW) and fish produced (WW) by Norwegian salmon farms in 2009. Production peaked from July to November, and there was a significant interannual variation in fish production, mainly dependent on the standing biomass of fish and the temperature during the year. On average, 67% of feed consumption and 68% of production took place in the summer–autumn period (June–November).

Production in individual counties ranged from 195 315 t in Nordland (NOR), which produced almost 20% of the salmon (1.02×10^6 t WW) in Norway in 2009, to 51 105 t in Finnmark (FIN), which accounted for 5% of total production. Other counties of importance for salmon production included Hordaland (HOR, 16%), Møre and Romsdal (MRO, 12%) and Sør-Trøndelag (STR, 12%).

The absolute daily growth rate of salmon for the different counties followed the same trend as the annual production (Fig. 3A). The mean FCR of fish farms (dry feed used per wet biomass of fish produced) in southern counties was slightly higher than in northern counties (Fig. 3B), though the differences were not statistically significant (ANOVA, $p = 0.053$). The mean economic FCR of Norwegian salmon farms in 2009 was 1.16 ± 0.08 .

The GE values showed small differences between the counties, with slightly higher yields in the northern counties (from STR to FIN; Fig. 3C,D). The differences between southern and northern counties were

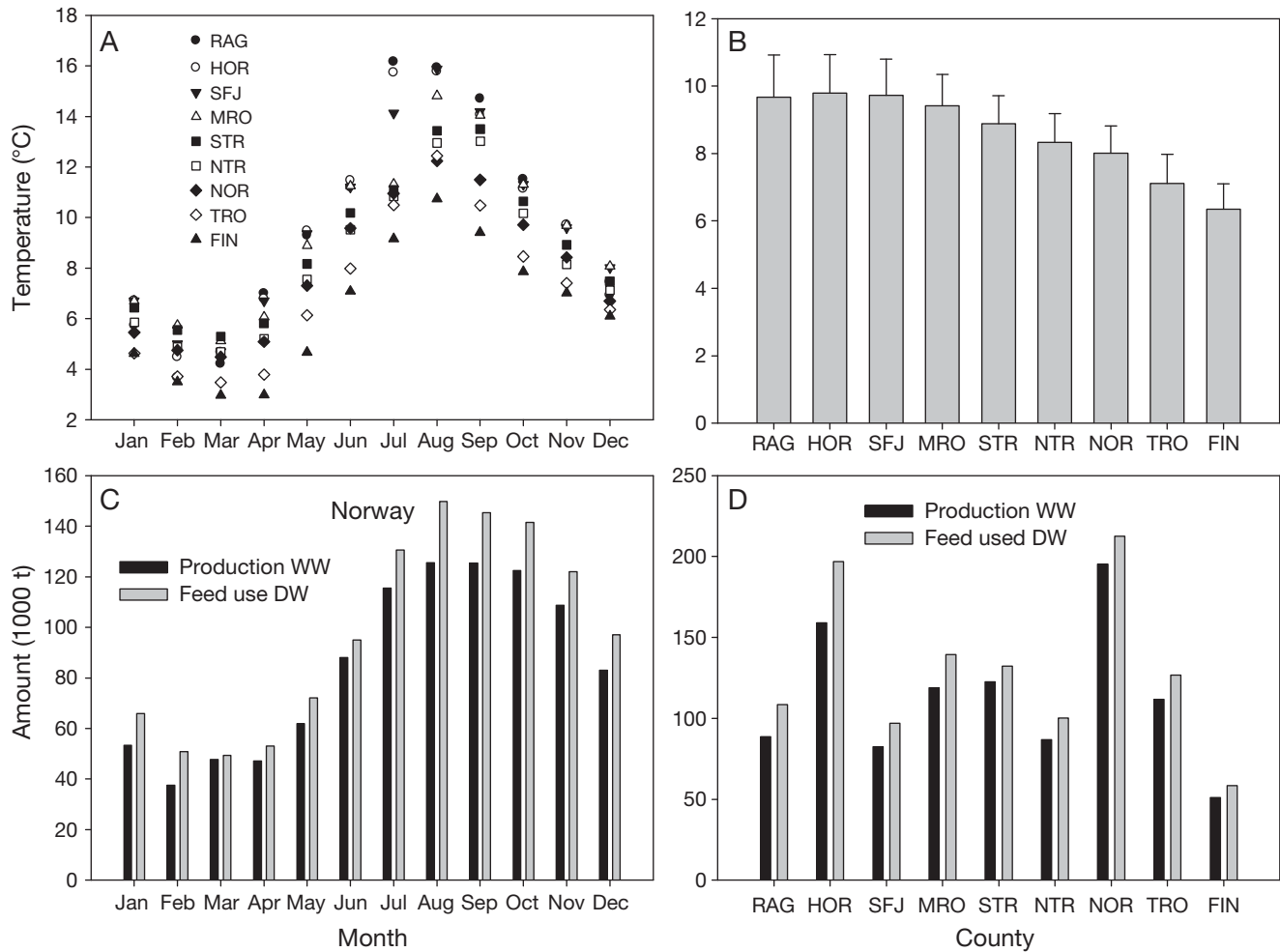


Fig. 2. Summary data on Norwegian salmon farms in 2009. (A) Monthly and (B) annual average surface water (0–15 m depth) temperature (°C), by county. (C) Monthly total salmon production (t wet weight, WW) and total feed supplied (t dry weight, DW), and (D) annual salmon production and feed supplied by county, based on Norwegian Fish Farmer Association (FHL) statistics. Counties are listed from south to north: top to bottom in (A), left to right in (B) and (D). RAG: Rogaland and Agder; HOR: Hordaland; SFJ: Sogn and Fjordane; MRO: Møre and Romsdal; STR: Sør-Trøndelag; NTR: Nord-Trøndelag; NOR: Nordland; TRO: Troms; FIN: Finnmark

statistically significant for GE_C and GE_P ($p < 0.05$), but not for GE_N ($p > 0.05$). The average GE_C , GE_N and GE_P values for all counties were 0.31 ± 0.01 , 0.39 ± 0.01 , and 0.31 ± 0.01 , respectively. This means that $31 \pm 1\%$, $39 \pm 1\%$ and $31 \pm 1\%$ of the consumed C, N and P, respectively, were retained in fish biomass. GE_N was higher than GE_C and GE_P , indicating that a lower proportion of consumed N than of C and P was released through defecation and excretion (or respiration).

Release rates of C, N and P components

The release rate of organic and inorganic C, N and P components followed the same trend and showed

a pronounced pattern of variation over the year (Fig. 4A–C), as did feed use and salmon production (Fig. 2C). The release rate of particulate, dissolved organic and dissolved inorganic wastes was highest in August with less than half of the maximum during the January–May period.

The model predictions were that the majority of the C and N waste from salmon farms was as respired CO_2 and excreted DIN, respectively, whereas the majority of the P waste was particulate. There was accordingly a higher proportion of feed C and N than of feed P released as inorganic molecules. Correspondingly, the fraction of particulate C and N released was lower than the fraction of particulate P (Fig. 4A–C).

The average DIN:DIP ratio of the waste over the year (Fig. 4D) was 14.7 ± 0.22 , well above the Red-

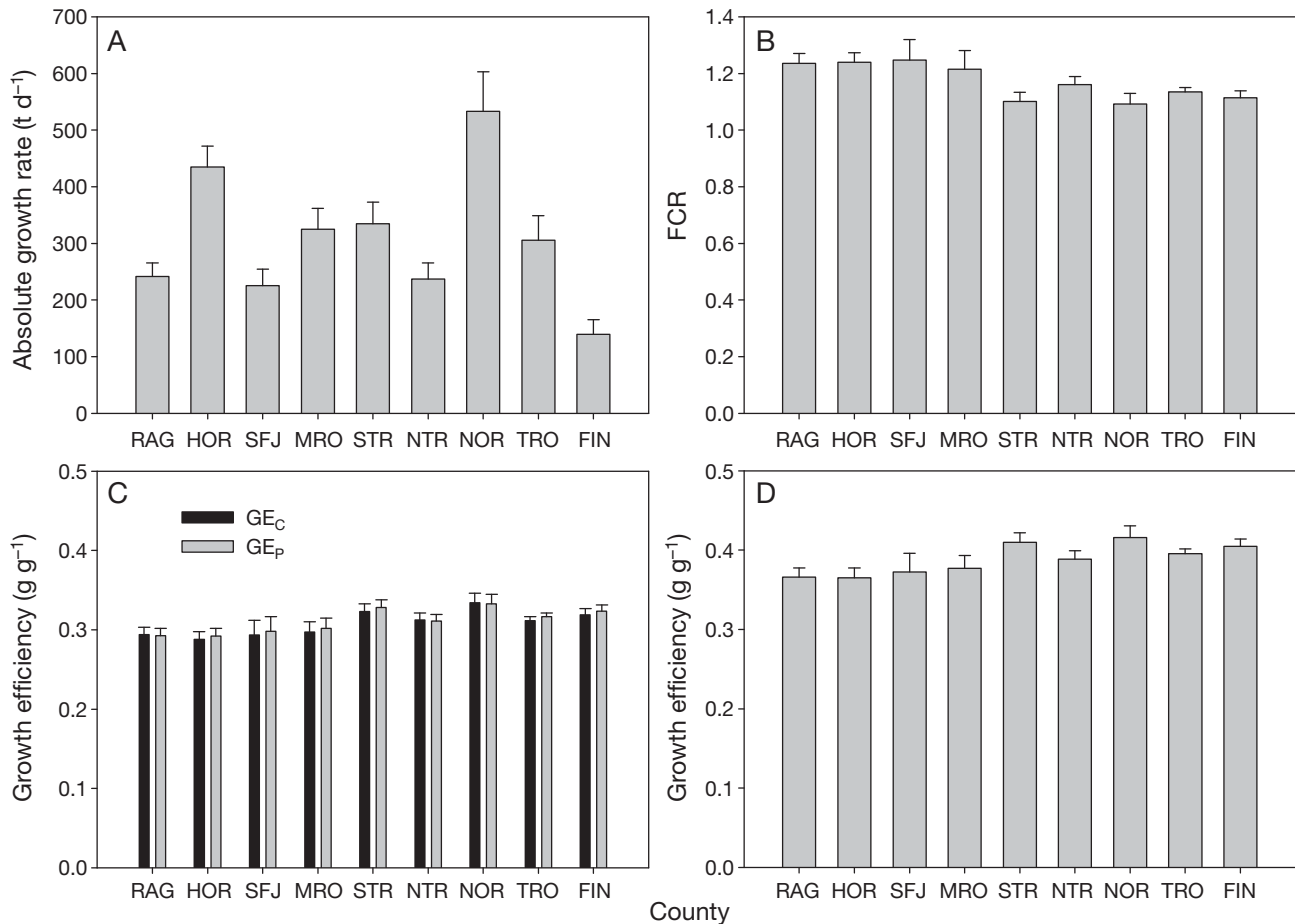


Fig. 3. Mass balance indices for Norwegian salmon farms, by county, 2009. (A) Absolute growth rate for salmon (mean \pm SE; $t d^{-1}$); (B) feed conversion ratio (FCR, dry weight feed per wet weight of fish produced). Growth efficiency (GE, g produced per g consumed, mean \pm SE) of (C) carbon (GE_C), phosphorus (GE_P) and (D) nitrogen (GE_N). Counties are listed from south to north, left to right; see Fig. 2 legend for full names of counties

field ratio of 7.2, suggesting that DIN was in excess of phytoplankton requirements relative to DIP. The model predicted a much lower and constant mean N:P ratio of 2.0 for both dissolved organic and for particulate organic waste. The average N:P ratio of total waste over the year was 5.4 ± 0.02 , not far from the Redfield ratio, but with P in excess.

Annual C, N and P mass balance of salmon farming

The total input of C, N and P (DW) through feeding in 2009 was 574 000, 81 000 and 13 500 t, respectively, and total salmon production was 1.02×10^6 t. Of the total input of feed, 70, 62 and 70% C, N and P, respectively, were released back to the environment as inorganic and organic waste, corresponding to 397, 50 and 9.3 kg C, N and P, respectively, t^{-1} WW of fish produced.

The carbon mass balance showed that 48% of total feed C was respired by the fish, 19% was released

through defecation and 30% was used for growth (Fig. 5A). Approximately 3% of the total feed input was resuspended from particles and became DOC.

The nitrogen mass balance (Fig. 5B) indicated that 38% of total feed N was incorporated and harvested as fish biomass, 45% was lost as DIN and 15% was released as PON. Approximately 3% of the total feed N was resuspended into the water as DON from particles, thus adding to the DON pool.

Of the total feed P, 44% of the input was released as POP, 30% was retained in fish, and 18% was lost as DIP. Approximately 8% of the total feed used was resuspended from particles to form DOP (Fig. 5C).

Analysis of model sensitivity

The model predictions depend on the input values of coefficients and variables, and Fig. 6 shows the effects of varying the fraction of feed losses (Fig. 6A,B), as-

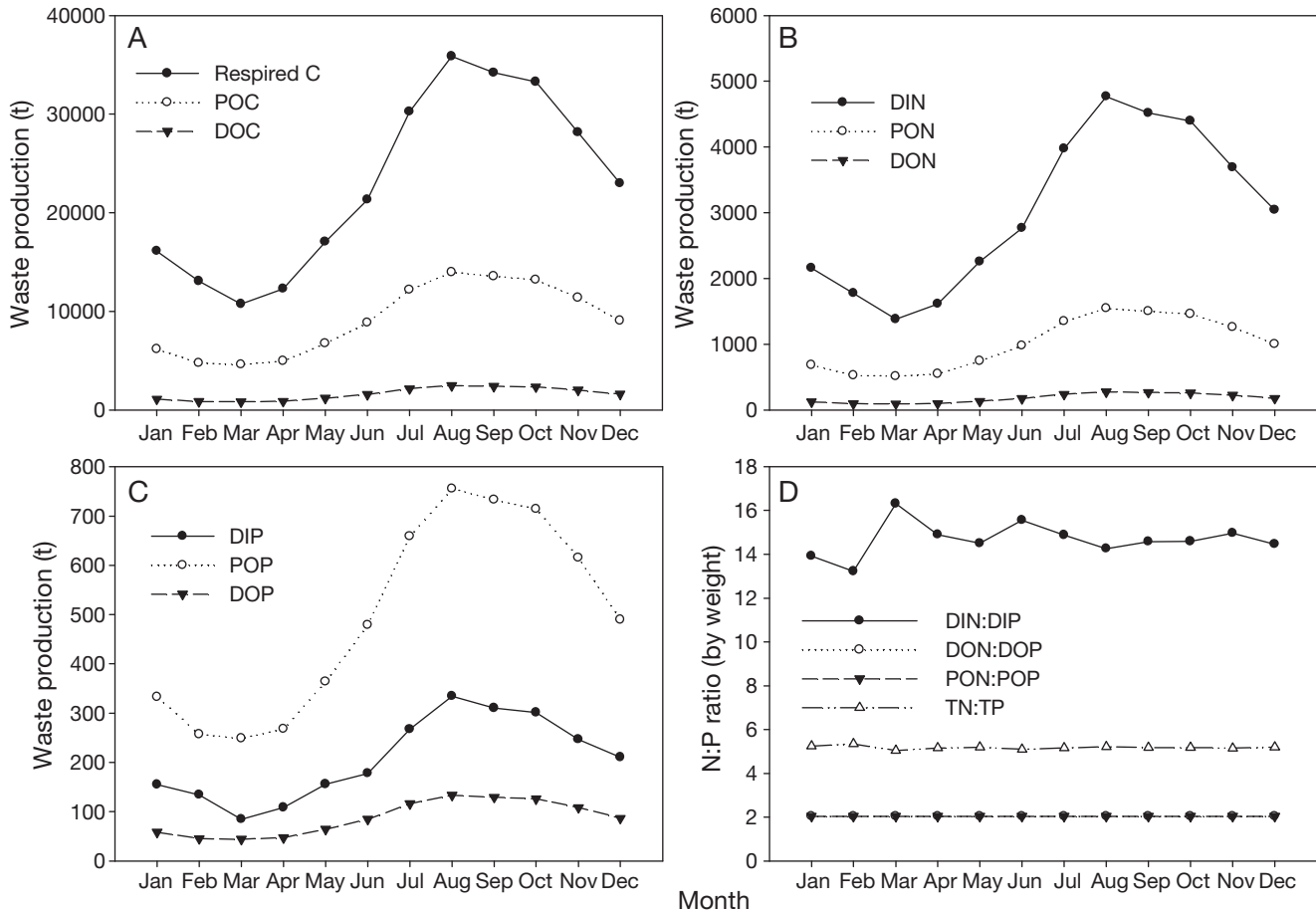


Fig. 4. Seasonal variation in C, N and P release rates (t mo^{-1}) and N:P ratios for nutrient components released from Norwegian salmon aquaculture in 2009. (A) respired CO_2 , POC and DOC; (B) DIN, PON and DON release; (C) DIP, DOP and POP release; (D) N:P ratios for different nutrient components. Note that DON:DOP and PON:POP ratio values overlap. TN, TP: total N and P, respectively; see Fig. 1 legend for other abbreviations

sumed to be 3% in the above calculations, and the AE (digestibility) of N (Fig. 6C) and P components (Fig. 6D) in the feed, which were assumed to be 85 and 50%, respectively, in the above calculations.

An increase in the loss of feed caused a decrease in percentage excretion of both DIN and DIP, and a similar increase in the percentage of combined organic waste. A minor variability in the feed loss from 0 to 5% did not cause major differences to model predictions.

A simulated variation in AE_N and AE_P affected the ratio of inorganic excretion to organic particulate defecation. The default value of 85% for AE_N used in the above calculations is believed to be fairly accurate (Berge et al. 1999, Opstvedt et al. 2003, Azevedo et al. 2011), but the value of AE_P is more uncertain, partly because of the wide variety of molecular P species that are incorporated in pellet feeds with a high agricultural product content (Cho & Bureau 2001,

Hua & Bureau 2006, Azevedo et al. 2011). The difference in the ratios of DIN:DIP and PON:POP (Fig. 4D) mainly originated in the lower AE_P value used in the calculations.

Production potential of salmon-driven IMTA

There is no accepted method for determining the proportion of dissolved inorganic and particulate organic aquaculture waste that can be used by co-cultured macroalgae and blue mussels, respectively, in open-water IMTA systems. Table 2 sets out the estimated potential of blue mussel wet flesh weight (WFW) and seaweed WW, and the area needed for that seaweed cultivation for the different counties (see 'Materials and methods'). The estimated potentials for the counties were closely related to the amount of feed used and fish produced. HOR showed the

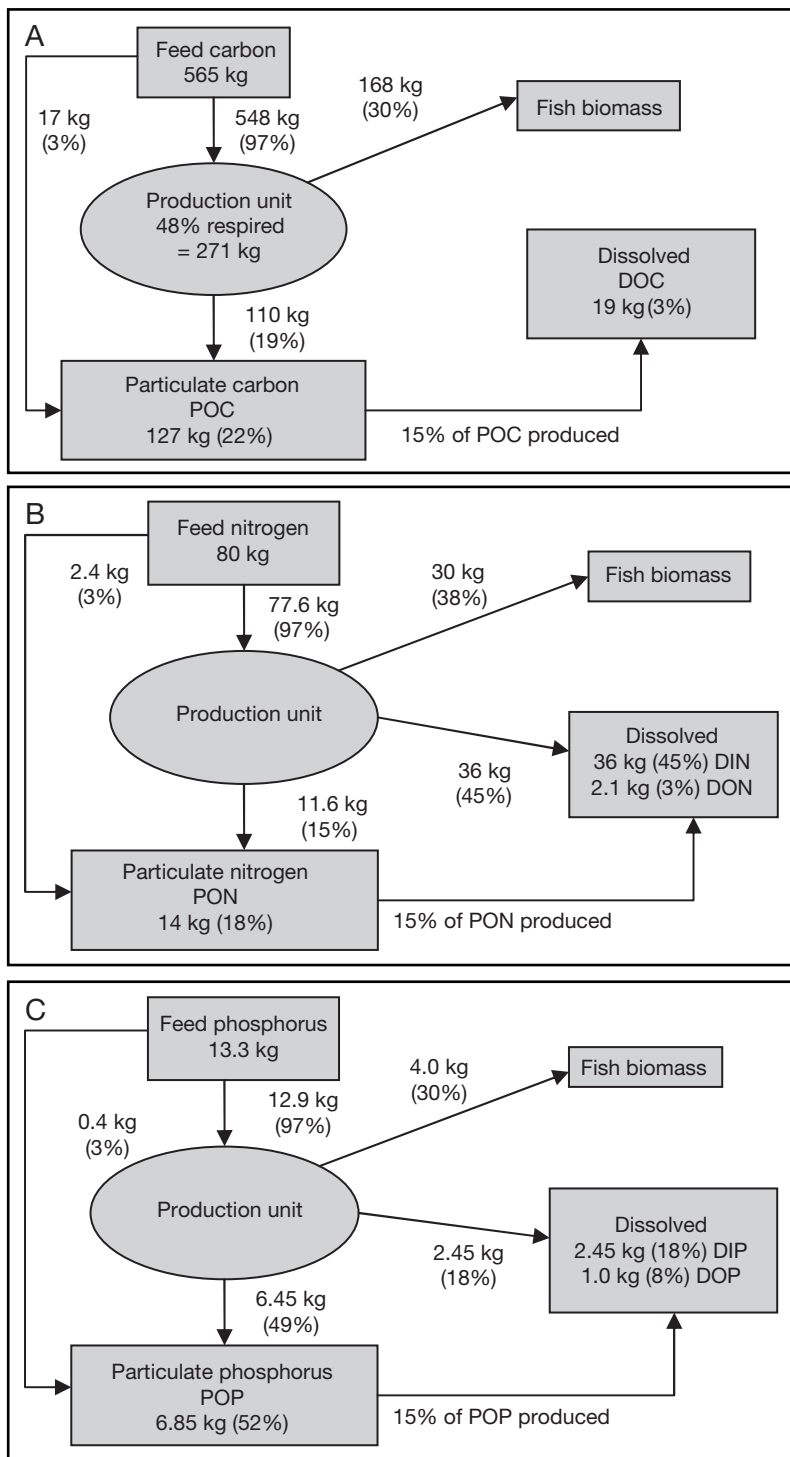


Fig. 5. Annual nutrient fluxes ($\text{kg} [\text{t produced} \times \text{yr}^{-1}]$) and components (kg t^{-1} produced) for Norwegian salmon farming in 2009: (A) C, (B) N and (C) P

highest potential for production of seaweed based on wastes from salmon farming, whereas NOR showed the highest potential for blue mussels. The lowest production potential was found in FIN. The potential

seaweed biomass (WW) for the whole country was in the range of 577×10^3 to $1730 \times 10^3 \text{ t yr}^{-1}$, while the potential mussel biomass (WW) was in the range of 7.2×10^3 to $22 \times 10^3 \text{ t yr}^{-1}$. The total DIN released from Norwegian salmon farming (not shown) corresponds to a seaweed production of around $5.8 \times 10^6 \text{ t yr}^{-1}$.

It is noteworthy that the potential for seaweed production was far higher than the potential for mussels. This is primarily a result of a smaller initial available resource and higher metabolic losses for mussels than for seaweeds. The area needed for seaweed cultivation for different counties followed the same trend as the potential seaweed biomass, and ranged from 4 to 44 km^2 for individual counties and from 82 to 247 km^2 for the country as a whole.

It is important to emphasize that the process of deriving production potential did not reveal accurate values that can be used for more detailed planning; however, it provides a preliminary estimation of the potential of IMTA driven by salmon farming in Norway. The fractions of wastes that can be acquired by seaweed and mussels are very uncertain, but the higher potential for seaweed than for mussels in salmon-driven IMTA, and the fact that seaweed cultivation requires a moderate area for growth, are both fairly robust conclusions.

Loading rate of nutrients

The potential impact on pelagic ecosystems of the inorganic nutrients excreted by farmed fish depends on the water volume to which these nutrients are drained. Table 3 summarizes our estimates of the volumetric loading rates of inorganic nutrients released from salmon farms by county in 2009 (see 'Materials and methods'). The annual

loading rate from aquaculture decreased steadily from the southern to the northern counties, with annual values ranging from 10 to 146 $\text{mg DIN m}^{-3} \text{ yr}^{-1}$ and 0.6 to 11 $\text{mg DIP m}^{-3} \text{ yr}^{-1}$.

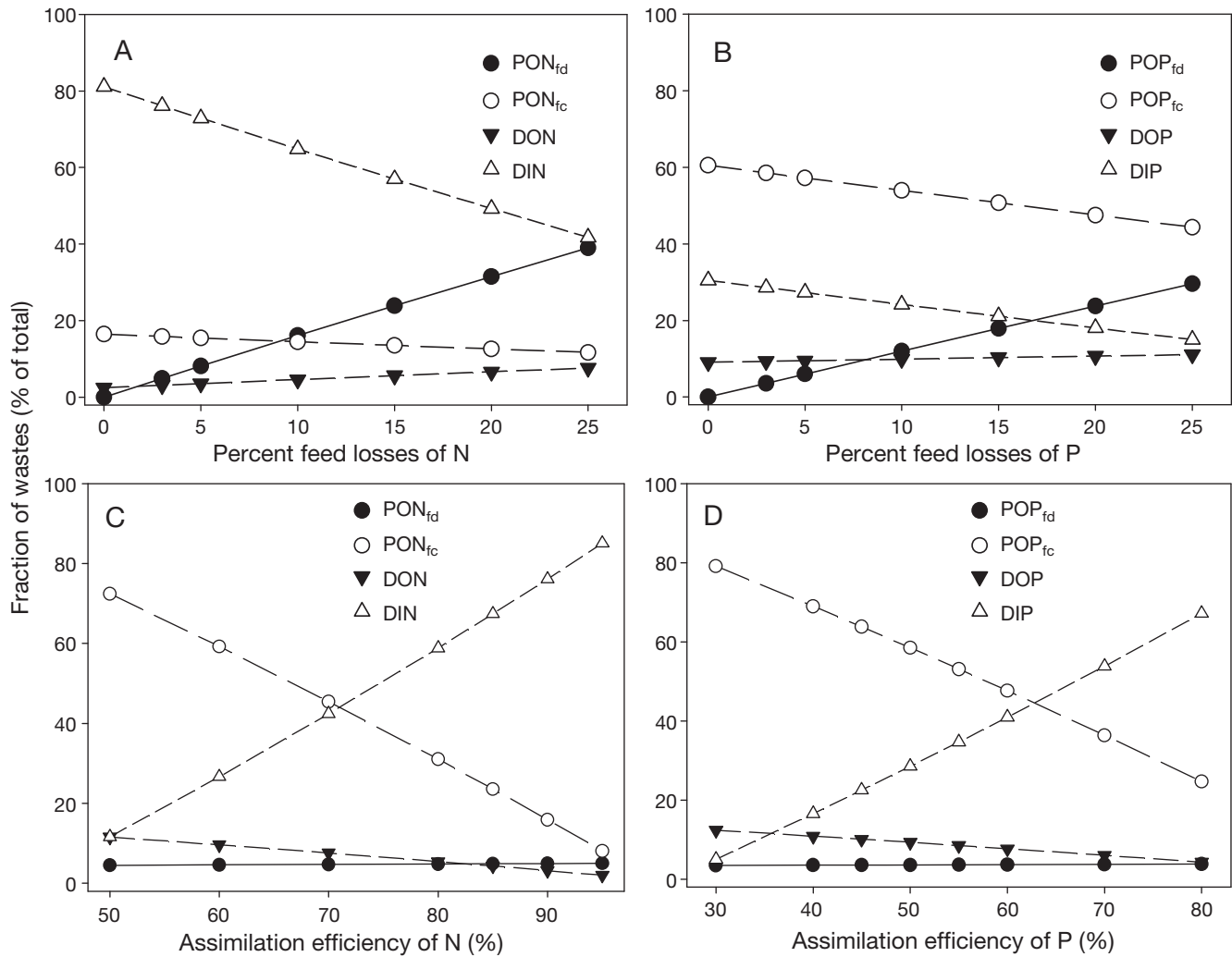


Fig. 6. Relative fractions of nitrogen (N) and phosphorus (P) waste components as a function of feed loss: (A) N, (B) P, and as a function of the assimilation efficiency of (C) N and (D) P. fc: faecal origin; fd: origin from feed; see Fig. 1 legend for other abbreviations

Table 2. Potential of Norwegian aquaculture for integrated multi-trophic aquaculture (IMTA). County-level breakdown of dissolved inorganic nitrogen (DIN) and particulate organic carbon (POC) available for assimilation by seaweed and blue mussels, respectively (assumed to be 10–30% of total DIN and 0.6–1.8% of total POC; see 'Materials and methods'), potential seaweed and blue mussel production, and the area needed for potential seaweed production. See Fig. 2 legend for full names of counties. WW: wet weight; WFW: wet flesh weight

County	DIN (t)	POC (t)	Potential seaweed biomass based on DIN (t WW)	Potential blue mussel biomass based on POC (t WFW)	Area needed for potential seaweed cultivation (km ²)
RAG	350–1060	60–180	56 000–168 000	660–1990	8–25
HOR	650–1940	110–330	102 000–307 000	1200–3610	15–44
SFJ	310–920	50–160	49 000–146 000	590–1780	7–21
MRO	440–1310	80–230	69 000–208 000	850–2560	10–30
STR	390–1160	70–220	61 000–184 000	810–2430	9–26
NTR	310–930	60–170	49 000–148 000	610–1840	7–21
NOR	630–1880	120–360	99 000–298 000	1300–3900	14–43
TRO	390–1160	70–210	61 000–184 000	780–2330	9–26
FIN	180–540	30–100	29 000–86 000	360–1070	4–12
Total			577 000–1 730 000	7200–22000	82–247

Table 3. County-level recipient volumes and corresponding volumetric loading rates of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) released by Norwegian salmon farms in 2009, showing annual totals and volumetric loading rates ($\mu\text{g m}^{-3} \text{d}^{-1}$) for the summer–autumn (June–November) and winter–spring (December–May) periods. See Fig. 2 legend for full names of counties

County	Area inside baseline (km^2)	Volume inside baseline (km^3)	DIN (t yr^{-1})	DIP (t yr^{-1})	DIN ($\text{mg m}^{-3} \text{yr}^{-1}$)	DIP ($\text{mg m}^{-3} \text{yr}^{-1}$)	DIN		DIP	
							($\mu\text{g m}^{-3} \text{d}^{-1}$) Summer–autumn (Jun–Nov)	($\mu\text{g m}^{-3} \text{d}^{-1}$) Winter–spring (Dec–May)	($\mu\text{g m}^{-3} \text{d}^{-1}$) Summer–autumn (Jun–Nov)	($\mu\text{g m}^{-3} \text{d}^{-1}$) Winter–spring (Dec–May)
RAG	1710	26	3532	252	138	10	471	283	34	20
HOR	2940	44	6449	464	146	11	501	300	36	22
SFJ	3490	52	3065	213	59	4.1	196	125	13	9
MRO	5220	78	4373	303	56	3.9	197	109	13	7.8
STR	6380	96	3862	249	40	2.6	140	81	9	5.5
NTR	4660	70	3113	213	45	3.1	166	78	11	5.4
NOR	30670	460	6263	407	14	0.9	53	22	3.5	1.4
TRO	9900	149	3872	261	26	1.8	102	41	6.9	2.7
FIN	12680	190	1797	122	9.5	0.6	40	12	2.8	0.8

Approximately 61 to 78% ($67 \pm 1.8\%$) of DIN and 62 to 78% ($66 \pm 2.2\%$) of DIP were released in the summer–autumn period in 2009 (June–November). In the summer–autumn period, the average daily loading rates varied between 40 and $501 \mu\text{g DIN m}^{-3} \text{d}^{-1}$ and between 2.8 and $36 \mu\text{g DIP m}^{-3} \text{d}^{-1}$. In the winter–spring period, the county-level nutrient loading rates varied between 12 and $300 \mu\text{g N m}^{-3} \text{d}^{-1}$ and between 0.8 and $22 \mu\text{g P m}^{-3} \text{d}^{-1}$. Nutrient loading rates related to sea surface area (m^{-2}) instead of volume were 15 times higher than the volumetric rates.

DISCUSSION

Intensive salmon aquaculture generates dissolved and particulate nutrient waste to the surrounding benthic and pelagic ecosystems (Mente et al. 2006), and considerable attention has been paid to the potential influence that organic matter and nutrients (biogenic waste) released from fish farming may have on coastal ecosystems (Sanz-Lázaro & Marín 2011). One main purpose of the present study was to evaluate the potential for IMTA driven by waste products from salmon farming in Norway. The different biogenic waste components are potential food resources for different farmed organisms at lower trophic levels, such as seaweed and mussels, and knowledge about both release rate and fate of biogenic waste components is paramount for deducing both IMTA potential and the effects that the waste may have on the ecosystems of receiving waters. IMTA is in fact a means to add value to investments in feed for aquaculture while simultaneously contributing to mitigation of potential negative impacts of fish farming on the environment.

Estimating the release rate of biogenic C, N and P waste from fish farms is important in the evaluation of the potential effects that salmon aquaculture may have on the environment and the potential of salmon-driven IMTA. We used a simple and transparent method to estimate the release rate of waste, which we suggest is also reasonably robust and flexible. The estimates of total release rate of C, N and P are dependent on statistical information about the use of feed and production of salmon, together with easily measurable coefficients describing the elemental C, N and P composition of feed and fish. The relative distribution of organic waste and dissolved inorganic waste is additionally dependent on the digestibility of C, N and P components of the feed and the rate of feed loss during feeding.

A fundamental question to be considered in using the mass balance method is whether or not the use of feed and the production of fish are relatively stable from month to month. Feed is used more or less continuously over time whereas fish production may to some extent be related to fish harvesting at the end of the life cycle. The events associated with harvesting might logically be more temporary and not as continuous over time as feeding. We found, however, that feeding and production (harvesting and biomass increment) are well balanced when FCR was calculated from month to month at the scale of a county (data not shown). The FCR values among counties were also stable (Fig. 3B).

The feed input is the most important input variable in the model. If the approximate value of FCR is adequately known—in the present case it might be assumed to be equal to the national average FCR for the Norwegian salmon industry—fish production can be estimated based on data for feed use divided

by the average FCR. This method is also appropriate for application to single fish farms where use of feed and harvesting are more variable and out of phase. In the present case, fish production estimates generated for single farms could use the average FCR for the county. However, calculations should always be made with care, and this procedure is only applicable for farms that apply a standardized pellet feed and modern feeding technology. Variable feed composition and digestibility should be reflected in the input coefficients used in the calculations (Table 1), and the coefficients may even be directly measured if necessary.

C, N and P release rates

Our calculations revealed that as much as 70, 62 and 70% of the total C, N and P, respectively, in feed input were lost to the environment in 2009 (Fig. 5A–C). This implies total annual C, N and P discharges of about 404 000, 50 600 and 9400 t of C, N and P, respectively. The monthly nutrient release rates over the year followed the same trend as feed use and fish production, as was also found in earlier studies (Davies 2000, Islam 2005). Approximately 67% of the total biogenic waste was released in the summer–autumn period (June–November) (Table 3). Moreover, of the total C input to the fish farms, 48% was respired as CO₂ and 19% was lost as POC (Fig. 5A). The majority of feed N consumed was excreted as DIN (45%), corresponding to 36 kg DIN t⁻¹ of salmon produced (Fig. 5B), in agreement with the DIN release rate by salmon farms in Scotland (35 to 45 kg N t⁻¹ fish produced; Davies 2000). The majority of feed P was lost as particles (44%, net POP) whereas 18% was released as DIP, corresponding to 2.45 kg DIP t⁻¹ of salmon produced (Fig. 5C).

Our estimates of nutrient release rates were similar to values obtained in some previous studies (Penczak et al. 1982, Phillips et al. 1985, Hall 1990, á Norði et al. 2011, Azevedo et al. 2011), but were lower than values reported by others (Gowen et al. 1985, Holby & Hall 1991, Hall et al. 1992, Chen 2000, Pérez 2002, Islam 2005, Corner et al. 2006). The selected values of input coefficients (Table 1) and other assumptions made affected the outcome of the predictions. As pointed out above, the stoichiometric C, N and P coefficients for feed and fish, the rate of feed loss and the digestibility of N and P components of the feed all affect the estimated release rates of the waste components in our calculations.

To illustrate the dependence of the predictions on a variable data input, we carried out some simple analysis of sensitivity of model predictions, following varying rate of feed loss and digestibility of N and P components of the feed. The total organic C waste (POC + DOC) increases and the release of DIC through respiration decreases for higher rates of feed loss, but the total release of N and P remains constant and independent of the rate of feed loss (Fig. 6A,B). The particulate N and P fraction of total loss, however, increases while the fraction of inorganic N and P decreases as feed loss increases (Azevedo et al. 2011). Our calculations revealed a similar release rate of DIN and total organic N for feed loss of approximately 15% while DIP was released in lower quantities than total organic P (POP + DOP) for all values of feed losses (Azevedo et al. 2011).

A similar pattern of variation was also apparent for a variable digestibility of N and P components of the feed (Fig. 6C,D). Reduced digestibility of N and P resulted in reduced relative release rates of DIN and DIP, respectively, and a similar increase in the release rate of total organic N and P. Because feed N is mainly associated with proteins, we assumed that 85% of the N is digested by fish, this being a normal value for fish feed protein digestibility, as declared by feed manufacturers (T. Lea, Skretting AS pers. comm.). The assumptions for N are fairly robust, whereas the digestibility of P components in salmonids feed is much more uncertain, partly because of the high diversity of molecular P species in the feed (Cho & Bureau 2001, Hua & Bureau 2006). Previous studies that reported the digestibility coefficients for P in rainbow trout have varied in the range of 0.43 to 0.71, depending on the methods used (Vandenberg & De La Noüe 2001, Bureau et al. 2003, Azevedo et al. 2011, Dalsgaard & Pedersen 2011). We assumed that 50% of the P in the feed is assimilated in salmon (Reid et al. 2009), to obtain approximate estimations of the released fractions of DIP and POP+DOP waste.

Evaluation of IMTA potential

Several macroalgae and filter or deposit feeding organisms have the potential to incorporate and utilize specific biogenic wastes from fish farms (Troell et al. 1997, Troell & Norberg 1998, Whitmarsh et al. 2006, Marinho-Soriano et al. 2009, Redmond et al. 2010, Abreu et al. 2011). There is, however, no general concept developed for evaluating appropriate plant and animal species or for estimating the specific IMTA potential of that species in a given aqua-

culture system and environment. As a first approach, and to illustrate the potential, we assumed that 10 to 30% of the total DIN and 0.6 to 1.8% of the total POC from salmon farms could, using appropriate technologies, be made available for cultured seaweed and blue mussels. Our exercise demonstrated a far higher potential for seaweed production, corresponding (by weight) to 60–170% of Norwegian salmon production, than for mussel, corresponding to 0.7–2.2% of national salmon production. The potential yield of mussels, based directly on small particles of feed and faeces, was relatively low, and would still be low even if all the small particles were made available for mussels. This suggested that particulate waste from salmon farming is less important than their natural food sources, even if the mussels are cultured close to the salmon farm.

As mentioned, these estimates are uncertain because many factors, such as the physical arrangement of the IMTA system, water currents, nutrient concentrations, light and temperature conditions, stocking densities, bio-fouling, and others, can limit nutrient uptake in seaweed and food consumption in blue mussels (Troell et al. 2003). In addition, the N and dry matter content in seaweed co-cultured with fish farm effluents can vary quite pronouncedly (Carmona et al. 2001, Porrello et al. 2003, Schuenhoff et al. 2003, Yang et al. 2006), which would affect our estimated potentials.

If, as assumed above, 10 to 30% of the released DIN is assimilated in macroalgae, it follows that 70 to 90% can become assimilated in phytoplankton, corresponding to 25×10^3 to 33×10^3 t of DIN (Table 3). This DIN will be widely spread by hydrodynamic forces and diluted to low levels, but can nevertheless support a realized phytoplankton biomass in the range of 1.3×10^6 to 1.6×10^6 t fresh weight (assuming a WW:N ratio of 50), which can in turn support a fresh biomass production of approximately 0.3×10^6 t of blue mussels, far higher than that supported by particulate waste (Table 2). However, this DIN resource is difficult to capture in IMTA, and will normally only stimulate the natural food webs.

Although the potential of producing co-cultured species in an open-water salmon-driven IMTA system cannot be accurately determined, our exercise was able to give a general idea of the economic potential of introducing an IMTA system. Practical testing of systems and locations are needed to draw final conclusions about the potential, but preliminary estimates such as ours may motivate that step and contribute to the growing interest in IMTA, as an ecosystem-based approach that can sustain added

value in aquaculture production, while mitigating waste output (Reid et al. 2009).

Evaluation of nutrient loading rates

The probability of harmful eutrophication in coastal waters should be evaluated based on estimation of the volumetric nutrient loading rate of the water system, defined as mass of nutrient input per water volume and time. A main modifying factor for ecological impacts is the capacity of the recipient waters to assimilate the nutrients, which is strongly related to water retention time and hydrodynamics (Yokoyama et al. 2004). This principle is very well established for freshwater systems, where the recipient volume and residence time of the water can be estimated relatively easily (Vollenweider 1976). No unique volume can be estimated for the receiving water masses for open coastal waters, but there is a need to define principles for doing so, in order to meet the requirements of developing and implementing an ecosystem-based management system for coastal waters (Tett 2008)

In Norway, county administrations will most likely become responsible for managing the national marine environment and resources, and we have estimated the volume for euphotic waters (mixed layers, 0 to 15 m depth) of the separate counties to a fairly good level of accuracy (Table 3). We suggest that these county-level water regions can be defined as the receiving ecosystems. From an ecological point of view, this pragmatic definition is as good as any other definition of the receiving ecosystem.

The ecological relevance of using a pragmatic county-level definition of receiving waters is a complex issue, but we suggest that the feasibility of this approach is mainly determined by the mixing efficiency of the euphotic water masses. Nutrients from most anthropogenic sources, among them aquaculture, are released from point sources. The suggested approach will work adequately and generate maximum values if the mixing time of the euphotic water that receives the point sources of nutrients is either faster or similar to the time needed for the organisms of the pelagic ecosystem to respond to an enhanced nutrient input. Phytoplankton will normally need 3 to 7 d to respond to a moderate increase in nutrient concentration, whereas crustacean zooplankton species may need >10 d (Buschmann et al. 2007, Olsen et al. 2007, 2011).

Water masses in relatively closed fjords may need a longer time for mixing, but it is also likely that some

outer and more exposed coastal waters may become well mixed within ~1 wk. The receiving water masses of exposed coastal waters will then exchange with coastal water outside the baseline from the very beginning, supporting the suggestion that estimated loading rates can represent a maximum value if mixing is fast, and a minimum if it is slower than 1 to 2 wk.

Three-dimensional hydrodynamic modelling may, at a later stage, support the estimation of mixing times needed for the different county-level water regions. Such information is relevant for appropriate localization of large fish farms and to minimize the ecological influence of nutrients as production further increases in the future (Tsagaraki et al. 2011).

Our exercise showed a gradual reduction in the volumetric loading rates of inorganic N and P from salmon farms from lower to higher latitudes along the Norwegian coast (Table 3), with lower N and P release rates through winter–spring than through summer–autumn seasons (Fig. 4B,C). The daily volumetric loading rates of DIN, which generally limit phytoplankton production in Norwegian coastal waters during summer–autumn, were estimated to range from 40 to 501 $\mu\text{g N m}^{-3} \text{d}^{-1}$ during June–November, with the lowest value for FIN and the highest for HOR (Table 3). The natural fertilization of nitrate (DIN) from deep water during the summer period (June–September) in central Norway was approximately 4 $\text{mg N m}^{-3} \text{d}^{-1}$ (Y. Olsen, O. Vadstein unpubl. data). This natural loading rate was 8 to 100 times higher than the DIN loading from aquaculture in the counties. Fertilization experiments (Olsen et al. unpubl., Vadstein et al. unpubl.) have also revealed that the pelagic ecosystem will not be negatively affected as long as the total anthropogenic supply of DIN does not exceed the natural supply rate to coastal waters by a factor of >1.3, suggesting that the current nutrient supply from aquaculture in the counties may be within safe limits.

CONCLUSIONS

The mass balance proposed in the present study is a reliable tool to estimate waste release rate on a national and regional scale, and this method is also appropriate for application to single fish farms. Our model demonstrated that 70, 62 and 70% of C, N and P, respectively, of the total feed input were released into the environment, equivalent to 404 000, 50 600 and 9400 t C, N and P, respectively. Consequently, the waste emission of C, N and P, respectively, was 397, 50 and 9.3 kg t^{-1} of fish produced, with total

salmon production of 1.02×10^6 t. Some 48% of feed C was respired as CO_2 , 45% of feed N excreted as DIN and 44% of feed P was released as solid waste. Our exercise of estimating a potential of IMTA driven by salmon farming demonstrated a far higher potential for seaweed production than for mussels. This was mostly because of smaller initial resource availability and higher metabolic losses for mussels than for seaweed. The daily volumetric loading rates of DIN from salmon farms were low compared to the natural loading rate of nitrate from deep water during the summer period (June–September) in central Norway, suggesting that the nutrient loading rate from aquaculture may remain within safe limits.

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