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ORIGINAL ARTICLE

## Regional impact from fin-fish farming in an intensive production area (Hardangerfjord, Norway)

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### Abstract

The Hardangerfjord is one of the largest salmon-farming areas in Norway, with an annual production of approximately 70,000 metric tonnes. The regional impact of fin-fish farming in a fjord environment was studied during 2008–2010. Ecological conditions in intertidal macroalgal and benthic deep basin communities were studied in addition to measurements of nutrients and chlorophyll-*a* values. Macroalgal communities in the intertidal zone and the deep water fauna communities showed a high ecological status in the intermediate part of the fjord and a good status in the inner part of the fjord. Faunal communities in the outermost basin indicate that the assimilative capacity for farm waste of this deep basin could be limited. Nutrients and chlorophyll-*a* values were within national thresholds defined as high water quality. The good ecological conditions of the parameters studied in the fjord show little evidence of a regional impact from the fish farming industry despite the intensive production level.

**Key words:** *Regional impact, benthic fauna, macroalgae, nutrients, fin-fish farming*

### Introduction

Salmon farming has grown rapidly in Norway since its onset in the 1970s and approximately one million metric tonnes are produced yearly. Waste from fish farming in open cages will have a large impact on the bottom communities in the area close to the farm and the release of nutrient and fine particulate material may also have a local influence in the shallow water communities in the near farm area. This local footprint of fish farming is well recognized and is subject to different types of monitoring in different areas in the world (Ervik et al. 1997, 2004; Hansen et al. 2001; Read & Fernandes 2003).

However, intense fish farming in an area may inflict regional impacts on marine ecosystems such as eutrophication, impact on shore communities and major changes in the environmental conditions for bottom communities. In order to develop environmentally sustainable fin-fish farming it is important to understand if such activities might have an impact beyond the immediate production area. The regional impact of intensive marine fish farming has been

little studied although some investigations do exist, mainly on the impact of dissolved nutrients (Gowen & Ezzi 1994; Soto & Norambuena 2004; Pitta et al. 2006; Kaymakci Basaran et al. 2010; Skejić et al. 2011).

The main fraction of the waste released during farming is bound in faeces and sinks rapidly ( $4\text{--}9\text{ cm s}^{-1}$ ) towards the sea bed (Chen et al. 2003). In low dynamic environments an increased flux of organic matter to the sea bed and co-occurring changes in infauna community structure are therefore mainly observed within 100 m of the farm's perimeter (Kutti et al. 2007b). Once settled, organic particles may be transported further in resuspension–deposition loops (Thomsen & Gust 2000), spreading the farm waste beyond the farm's perimeter. In western Norwegian fjords, near-bed current speeds above the threshold at which salmon faecal pellets are resuspended, i.e.  $10\text{ cm s}^{-1}$  (Cromey et al. 2002), occur episodically in connection with intermediate water exchange (density-driven current) and episodic deep-water inflows (Aure et al. 2007). In Bjørnafjord

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(Norway), farm waste was traced and signs of a moderately enriched infauna community were observed in an accumulation area 1 km away from a fish farm (Kutti et al. 2007a, 2007b). In intensive salmon farming areas where several farms have been active in the same body of water for an extended period of time, wider impacts of the release of organic waste should be expected.

Norwegian coastal waters are normally nitrogen-limited in the euphotic zone during summer (Aure & Johannessen 1997). Salmon growth is usually highest at this time of year and the emissions of nutrients will thus be high when nitrogen is naturally limited. An addition of dissolved nitrogen can stimulate phytoplankton growth and plankton blooms (Gowen et al. 1992; Boynton et al. 1996; Pedersen & Borum 1996; Bricker et al. 2003). If the production is not recirculated in the euphotic zone but settles on the sea bed, increased organic loading may occur and in extreme situations oxygen depletion in the basin water (Gowen & Bradbury 1987; Best et al. 2007). Nitrogen addition can also change the seaweed communities in the littoral zone by stimulating growth of annual, rapidly growing species which might out-compete perennial habitat-building species and cause shifts from highly diverse macroalgal communities dominated by perennial brown algae to low-diversity communities dominated by opportunists and annual species (Rueness & Fredriksen 1991; Bokn et al. 1992; Munda 1996; Pihl et al. 1999; Worm & Sommer 2000; Krause-Jensen et al. 2007).

The European Water Framework Directive (WFD, 2000/60/EC) establishes a framework for monitoring and, when required, the improvement of European coastal waters. WFD is incorporated into the Norwegian Water Management Regulations that have an objective of reaching at least a good status for all Norwegian waters by 2021. The assessment of the ecological quality within coastal waters is based on a combination of both biological and physical-chemical quality elements. The biological elements include assessments of phytoplankton biomass (i.e. concentrations of chlorophyll-*a*), macroalgae (maximum depth distribution of selected species and diversity in the littoral algal communities) and soft bottom fauna (i.e. diversity and NQI1 (Norwegian Quality Index 1)). The biological quality elements are considered as the most important indicators for determining water quality, as they are a direct measurement of the impact from pollution on organisms and may also indicate the long-term effects of pollutants in ecosystems, even when values beyond thresholds are difficult to detect (Bermejo et al. 2012). Development, calibration and testing of tools and criteria for different water

types in Norwegian coastal waters are still in progress.

The objective of this study is to assess regional impacts of intensive salmon farming by a classical monitoring programme of nutrients and chlorophyll-*a* values, but also by the use of novel tools (benthic bottom fauna and macroalgal communities) provided by the European Water Framework Directive (WFD, 2000/60/EC).

## Material and methods

### *The study area*

The Hardangerfjord is the second largest fjord in Norway, penetrating 179 km inland in a north-easterly direction, and it is the third largest fjord in the world (Figure 1). The sea surface area inside the main sill is approximately 800 km<sup>2</sup>. The main sill is 150–200 m deep and is situated near the island of Huglo as indicated in Figure 1, and the depth increases inwards in the fjord, reaching the greatest depth of 890 m in the inner basin of the fjord inside Vikingnes. Previous studies estimate a natural sedimentation rate of 5 mm per day in the inner area, mainly of fine material discharged by rivers with a high content of silt (30–80%; Holtedahl 1975). Water exchange in the surface layers of the fjord is high due to tidal waves, intermediate water exchange and fresh water run-off into the fjord. Near-bed current speeds may thus be low in many areas (Aure et al. 2007; Asplin et al. 2014). The retention time for surface water in the fjord is approximately one month (Ervik et al. 2008). Freshwater discharge from the drainage area, the Folgefonna glacier and rivers create a brackish surface layer throughout the fjord mainly during summer after snow melting. The inner fjord branches experience a brackish surface layer (salinity  $\leq 25$ ) in all seasons.

The Hardangerfjord is one of the most intensively used salmon-farming areas in the world, with an annual production of approximately 70,000 metric tonnes (Taranger et al. 2011). The production takes place in open net-cages and dissolved nutrients and organic wastes are released directly into the surrounding environment. Calculated using the Ancylus-MOM Fish-model (Stigebrandt et al. 2004), 127 metric tonnes of DIP (dissolved inorganic phosphorous) and 280 tonnes of POP (particulate organic phosphorous) from the production are released into the environment each year. Estimated annual nitrogen emissions are 770 metric tonnes of DIN (dissolved inorganic nitrogen) and 1756 metric tonnes of PON (particulate organic nitrogen). The production of 70,000 metric tonnes of salmon will produce around 7000 metric tonnes of particulate

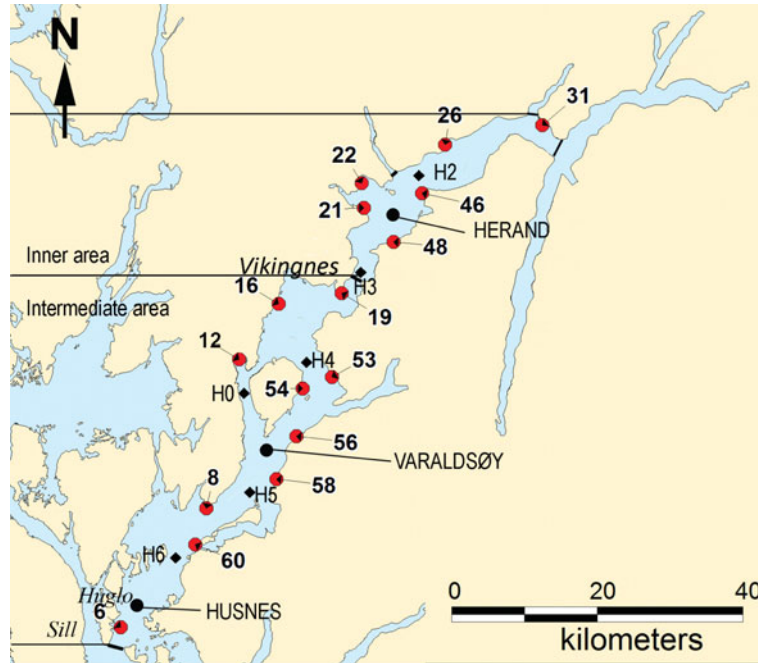


Figure 1. Map of the Hardangerfjord. Sites investigated in the period 2008–2010 are shown as: diamonds (H), nutrients and CTD; red (grey) circles (number), intertidal macroalgae sites; black circles (Herand, Varaldsøy, Husnes), soft bottom infauna sites. Map: Vidar Wennevik.

organic waste and contribute substantially to the total annual flux of organic matter to the infauna communities (Kutti 2008).

Most of the fish production takes place in the intermediate and outer part of the fjord, while the inner part of the fjord harbours fewer farms. In the intermediate and outer area the fish production is the main contributor to nutrient enrichment, while in the inner area the main contributor is natural runoff from land (Anon. 2011). The maximum allowed fish biomass in the intermediate area is high relative to the sea surface area (190 metric tonnes of fish/km<sup>2</sup>), while in the inner area there is relatively low biomass of fish compared to the sea surface area (56 metric tonnes of salmonid fish/km<sup>2</sup>; Norwegian Directorate of Fisheries). The annual production of fish is approximately 10–20% lower than the maximum allowed biomass because not all farms are in operation at the same time. The fish farms are generally situated 100–300 m from land along the steep hard rock shores of the fjord. In the fjord branches there is no fish production and these are not included in this study; nor is the outer area of the fjord, where we find a more coastal dynamic environment considered to be less sensitive to effluents from fish farming. In this study we have focused on the inner and intermediate area of the fjord (Figure 1; detailed information on sampling sites is given in Supplementary Material A). The intermediate area has a rather narrow entrance to the inner area and the deep basins of the two areas

are partly separated by a deep sill at approximately 500 m depth, close to Vikingnes (Figure 1).

#### Nutrients and fluorescence

Six locations, two in the inner area and four in the intermediate area, were selected for sampling of physico-chemical parameters (Figure 1). The sites were situated at least 500 m away the shore and more than 1 km from fish farms to ensure that the water was representative of the fjord water and not influenced by local emissions of nutrients. Water samples were collected from depths of 0, 2, 5, 7 and 10 m approximately monthly in the period March 2008–August 2010. Salinity, temperature and fluorescence (chlorophyll-*a*) were measured by a CTD (SD 200 W, SAIV A/S). Measurements were recorded every second while the CTD was lowered at a rate of <math>0.5 \text{ m s}^{-1}</math> from the surface to 30 m depth. Water samples were stored in a cool, dark area and taken to the laboratory and were principally analysed for nitrate, nitrite, phosphate and silicate within 24 h. In some cases when this was not possible, the nutrient samples were immediately fixed with 0.02 ml chloroform for later analysis. Water samples for the analysis of total N and P were taken in February 2009 and monthly in the period June to August 2010. Dissolved inorganic nutrients were analysed according to standard methods (Parsons et al. 1992). Nutrient values were calculated as mean values for each station in surface water at 0–10 m depth for the summer months

(June–August) and winter months (December–February). Chlorophyll-*a* values were calculated as mean values for each station in surface waters (0–10 m) during summer months (June–August). We have used the national thresholds for nutrient and chlorophyll-*a* values given by the Norwegian Climate and Pollution Directorate (Molvær et al. 1997) for determining water quality with these parameters.

#### *Intertidal macroalgal communities*

To assess the condition of the quality element of macroalgae required by the Water Framework Directive, we have used methods described in Guidelines for classification of ecological quality in water, which are still under development for Norwegian waters (Anon. 2009).

The multimetric index for macroalgae communities gives a value for normalized shore diversity (reference condition), which gives the characteristics for the macroalgal communities on a pristine shore. The normalized shore diversity is a list of species that is commonly associated with undisturbed conditions (reduced species list, RSL) and is specific for a geographic region. The water type in Hardangerfjord is characterized as Ns4A (strongly freshwater-influenced fjord, North Sea, Lindesnes–Korsfjorden). The multimetric index for macroalgal intertidal communities is only developed and validated for the water types Ns1B, and Ns2B and is suggested for the water types No1A and No1B, which are the areas north of Korsfjorden on the Norwegian west coast (Anon. 2009). Currently there is no available index for waters that are strongly influenced by fresh water run-off such as fjords. With the lack of such a fjord index we have chosen to use the available index to assess the condition of the macroalgal communities in the fjord environment.

Data for evaluation of the environmental condition of the macroalgal communities at 16 sites in Hardangerfjord (Figure 1) were compiled from the investigations of the macroalgal communities in the fjord during the summer of 2008–2009 (Husa et al. 2014). The sites were not randomly selected because this study was a reinvestigation of sites that were studied in the period 1955–1960 (Jorde & Klavestad 1963). Macroalgal communities in the intertidal and shallow subtidal zone were studied at 6 sites in the inner area and 10 sites in the intermediate area of the fjord during low tide in June 2008 and 2009. The sites were mainly situated on protruding land with a rocky substrate and the closest distance to a fish farm was 800 m and as such they should be characteristic of the intertidal macroalgal community in the fjord.

Normalized shore diversity was calculated for each site based on a scoring system evaluating parameters such as substrate, grazers, filter feeders and habitat complexity of the shore. An ecological quality ratio (EQR value) for the combined parameters (1) proportion of Chlorophyta taxa, (2) proportion of Rhodophyta taxa, (3) proportion of opportunistic taxa, (4) the ratio between late successional or perennial taxa (ESG 1 species) and opportunistic or annual taxa (ESG 2 species) at each individual site was calculated as described in the guidelines (Anon. 2009).

#### *Soft bottom infauna and oxygen*

Three sites were selected for deep-water infauna sampling in Hardangerfjord: Herand (837 m deep), Varaldsøy (642 m deep) and Husnes (455 m deep) (Figure 1). The three sites were situated at least 3 km from any farms in separate deep basins distributed in the inner and intermediate part of the fjord. At all sites five replicate grab samples of infauna (> 1 mm) were collected using a 0.1-m<sup>2</sup> van Veen grab, following the international standards given for soft bottom surveys (ISO 16665:2005). On board, the samples were labelled and preserved in 4% formaldehyde. In the laboratory the fauna samples were sorted and transferred to 70% ethanol. The infauna was identified to the lowest possible taxonomic level and counted. Before analysis the species list was checked to remove species that are not quantitatively sampled with the grab, i.e. pelagic organisms and highly mobile fauna. Infauna abundance (N), total number of species (S), diversity (H'), AZTI Marine Biotic Index (AMBI; Borja et al. 2003) and Norwegian Quality Index 1 (NQI1; Anon. 2009) were calculated for each sample separately. The AMBI values were calculated using the M-AMBI software (<http://ambi.azti.es/index.php?lang=en>). The NQI1 values were calculated according to the following formula:

$$NQI1 = [0.5 \times (1 - AMBI/7) + 0.5 \times (SN/2.7) \times (N/(N + 5))]$$

where SN is  $\ln S / \ln(\ln N)$  (Anon. 2009).

The average values for H' and NQI1 of the five replicate samples were used to assess the ecological status of the sites. N, S and the AMBI value are not used in the classification but only in the calculation of NQI1. As there are no fjords without fish farms, the ecological status was assessed against the standard reference conditions for Norwegian coastal waters, i.e. 0.78 for NQI1 and 4.4 for H' (Anon. 2009), corresponding to undisturbed pristine conditions (Borja et al. 2004). Thus, EQRs for NQI1 were created by dividing the present value by the reference value of 0.78. The thresholds for each of

the ecological status levels for H were:  $>0.86$  = high,  $0.68$ – $0.86$  = good,  $0.43$ – $0.68$  = moderate,  $0.2$ – $0.43$  = poor,  $<0.2$  = bad and for NQI1:  $>0.92$  = high,  $0.81$ – $0.92$  = good,  $0.63$ – $0.81$  = moderate,  $0.4$ – $0.63$  = poor,  $<0.4$  = bad (Anon. 2009). Since most inter-calibration work has been focused on the NQI1, main focus should be given to this element when evaluating the ecological status of a given body of water in Norway (Anon. 2009). Oxygen saturation in the near-bottom layer was measured in the deep basin of the inner area of Hardangerfjord at Herand and in the intermediate area at Husnesfjorden using a CTD with an oxygen sensor and Winkler titration, respectively.

## Results

### *Chlorophyll-a and nutrient values*

Nutrient values and chlorophyll-*a* were within the thresholds of a high water quality according to national guidelines (Figure 2) at all six sites in the inner and intermediate area of the Hardangerfjord. Chlorophyll-*a* values typically reached levels of  $3$ – $6 \mu\text{g l}^{-1}$  during spring bloom with a peak in March (Figure 2). The highest mean values of chlorophyll-*a* were detected at station H0 and station H5 during spring bloom. Mean summer values at all sites ranged from  $0.97$  to  $1.15 \mu\text{g l}^{-1}$ , which is within the threshold indicating a high quality (threshold high quality  $<2.0 \mu\text{g l}^{-1}$ ) (Figure 2).

Nitrate values at all sites were low during summer, varying from not detectable to  $0.5 \mu\text{mol l}^{-1}$  (threshold high quality  $<0.85 \mu\text{mol l}^{-1}$ ). During winter, nitrate levels reached a maximum value of  $6.5 \mu\text{mol l}^{-1}$  in February, just before the spring bloom (threshold high quality  $<6.43 \mu\text{mol l}^{-1}$ ). Phosphate summer values in the fjord were mainly within the threshold of high water quality (threshold high quality  $<0.13 \mu\text{mol l}^{-1}$ ), with the exception of measurements in June 2009, where the values were slightly higher ( $0.14$ – $0.16 \mu\text{mol l}^{-1}$ ) at all sites. Winter mean values of phosphate reached a maximum of  $0.40 \mu\text{mol l}^{-1}$  in February (threshold high quality  $<0.52 \mu\text{mol l}^{-1}$ ). At the innermost station in H2 the phosphate values remained high, with a maximum of  $0.44 \mu\text{mol l}^{-1}$  also during March and April in 2009 (Figure 2).

Winter values (only one measurement in February 2009) of Total P varied between  $0.53$  and  $0.59 \mu\text{mol l}^{-1}$  (threshold high quality  $<0.68 \mu\text{mol l}^{-1}$ ) and the corresponding Total N values varied between  $12.9$  and  $14.2 \mu\text{mol l}^{-1}$ , with the highest concentration at the outermost station in the fjord (threshold high quality  $<21.07 \mu\text{mol l}^{-1}$ ). Summer values measured from June 2010 to August 2010

showed Total P values between  $0.27$  and  $0.40 \mu\text{mol l}^{-1}$  (threshold high quality  $<0.39 \mu\text{mol l}^{-1}$ ) and the corresponding Total N values ranged between  $8.9$  and  $13.6 \mu\text{mol l}^{-1}$  (threshold high quality  $<17.86 \mu\text{mol l}^{-1}$ ) and both were within the national threshold of high water quality.

### *Macroalgae*

The nine macroalgal sites examined in the intermediate part of the fjord all showed a high ecological status (Table I). The number of species and opportunistic species varied little between sites and a high abundance of opportunistic species was not found at any of these sites. The littoral zone in this area was dominated by the fucoids *Ascophyllum nodosum* (Linnaeus) Le Jolis, *Fucus vesiculosus* Linnaeus and *Fucus serratus* Linnaeus, with a narrow zone of *Fucus spiralis* Linnaeus at most sites. The supralittoral brown alga *Pelvetia canaliculata* (Linnaeus) Decaisne & Thuret appeared scattered at four of the sites. The kelp *Saccharina latissima* (Linnaeus) C.E. Lane, C. Mayes, Druehl & G.W. Saunders was common in the shallow subtidal at all sites except from Haukanes (54) in this area and *Laminaria digitata* (Hudson) J.V. Lamouroux was common in the shallow subtidal zone at six sites (a detailed species list (RSL) is to be found in Supplementary Material B).

Macroalgal communities at five of the six sites examined in the inner area of the fjord were classified as being of a high quality ecological status, while the innermost site (31) was classified as being of a good ecological status (Table I). Fewer species were recorded at the two innermost sites than at the four sites located further out in the fjord (Table I). Site 31 had a particularly low species number as it was almost completely dominated by blue mussels (*Mytilus edulis* Linnaeus, 1758). The dominant brown algae at the innermost sites were *A. nodosum*, *F. vesiculosus* and *F. serratus*. The fucoid *F. spiralis* was recorded as common at the two outermost sites, but was only sporadically found at the innermost sites. The supralittoral brown alga *P. canaliculata* was not present in the inner area of the fjord. The kelps *L. digitata* and *S. latissima* were more scarce in this area, but were common in the shallow subtidal zone at sites 26 and 46 (detailed species list (RSL) in Supplementary Material B).

### *Soft bottom infauna and oxygen*

In the inner basin of Hardangerfjord infauna samples were collected at one site only, at 837 m depth at Herand, the deepest part of the fjord (Figure 1). This site was characterized by a rather low infauna

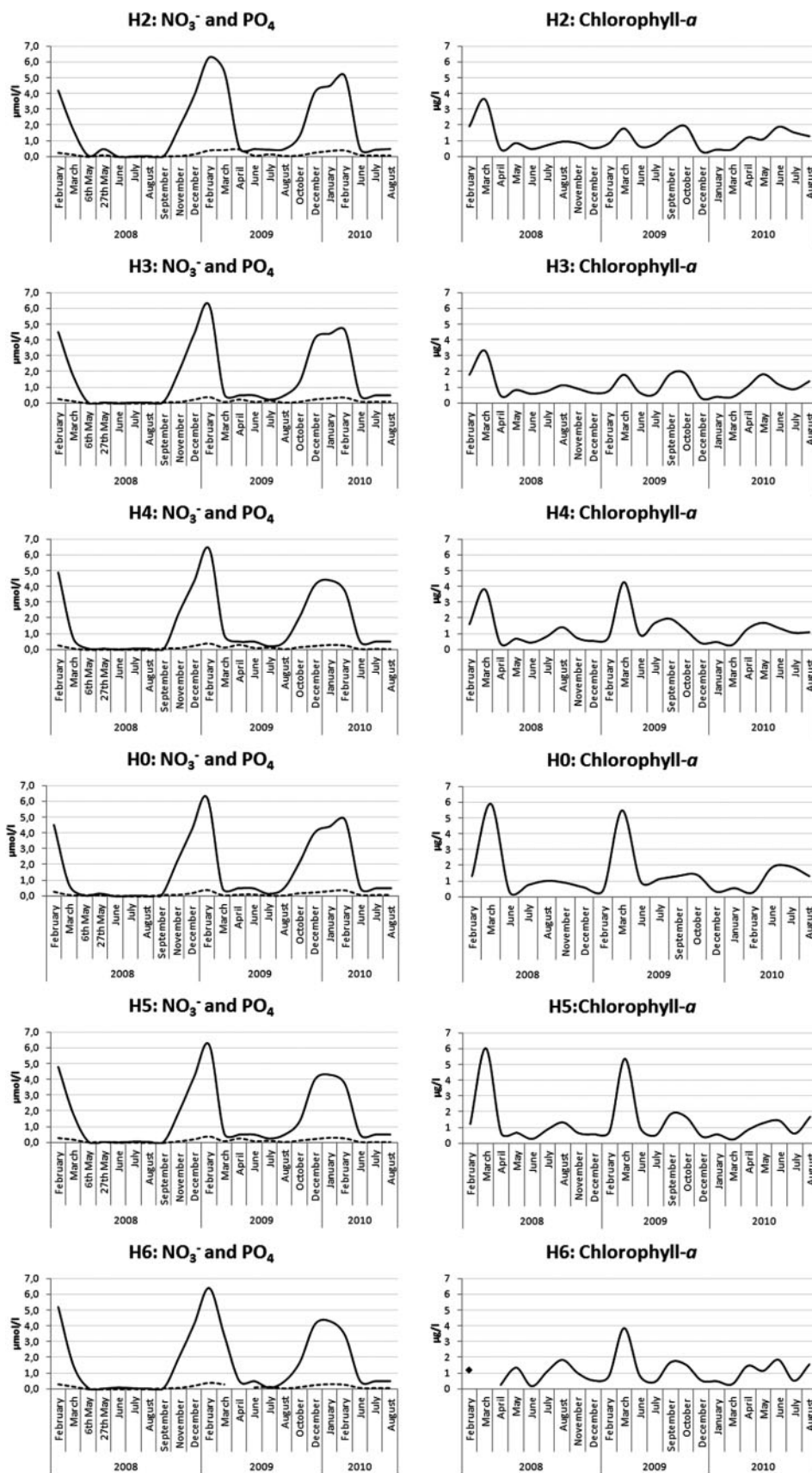


Figure 2. Mean nutrient and chlorophyll-*a* values measured in surface water (0–10 m depth) in Hardangerfjord in the period 2008–2010. On left half of figure, continuous line =  $\text{NO}_3^-$ , dashed line =  $\text{PO}_4$ . Site location, see Figure 1.

Table I. Species richness and calculated EQR values (ecological quality ratio) showing ecological status for the biological quality element *intertidal macroalgae* studied at 16 sites in Hardangerfjord during the summers of 2008 and 2009.

	Locality	Number of Rhodophyta	Number of Chlorophyta	Number of opportunists	Species richness	Normalized shore diversity	EQR	Status
Inner area	31 Nøstflot	7	1	2	14	1.36	0.788	Good
	26 Ålvik	9	1	4	21	1.72	0.856	High
	46 Samlaneset	13	4	6	28	1.44	0.830	High
	22 Aknes	13	3	5	25	1.58	0.837	High
	21 Øystese	11	4	6	24	1.36	0.815	High
	48 Solenes	13	3	5	25	1.51	0.845	High
Intermediate area	19 Ljonestangen	12	4	6	27	1.00	0.810	High
	16 Skjerring	14	3	5	28	0.80	0.810	High
	54 Haukanes	13	3	5	29	1.14	0.845	High
	53 Apalnes	11	3	5	26	1.14	0.830	High
	56 Svoldal	11	3	6	26	1.14	0.822	High
	12 Mundheim	13	4	6	31	1.00	0.828	High
	58 Løfallstrand	13	5	8	33	0.93	0.812	High
	8 Steinesnes	14	3	5	27	0.87	0.811	High
	60 Skorpa	11	5	7	30	1.14	0.801	High
	6 Huglo	13	3	4	27	1.14	0.834	High

abundance ( $67 \pm 16$  SD) and good species diversity ( $3.42 \pm 0.26$  SD). The infauna community at the site was dominated by the errant polychaete *Paradiopatra fiordica* (Fauchald, 1974) and juveniles of the bivalve *Thyasira* sp. (Table II). The average EQR for NQI1 was 0.92, placing the site just at the boundary between a high and a good ecological status (Table III). The average EQR for H' was 0.78 and the oxygen concentration of the bottom water was  $4.17 \text{ ml l}^{-1}$ , both supporting the classification of the site into a good ecological status.

In the intermediate basin of Hardangerfjord infauna samples were collected at two sites, just south of the island of Varaldsøy at a depth of 642 m and outside Husnes at a depth of 455 m (Figure 1). The Varaldsøy site was characterized by a rather high infauna abundance ( $116 \pm 50$  SD) and high infauna diversity ( $4.20 \pm 0.14$  SD). The bivalve *Kelliella* sp., the polychaetes *Paradiopatra fiordica* (Fauchald, 1974) and *Chaetozone setosa* Malmgren, 1867 and the ophiuroid *Amphilepis norvegica* (Ljungman, 1865) were dominant in all five grab samples (Table II). The average EQR for NQI1 for the Varaldsøy site was 0.96, indicating a high ecological status of the site (Table III). Average EQR for H' was 0.95, arriving at the same classification. The Husnes site

was characterized by an abundance of infauna similar to that found in Varaldsøy ( $117 \pm 46$  SD) and a diversity of  $4.0 \pm 0.48$  SD (Table III). The infauna community at the site was dominated by the cirratulid polychaetes *Aphalochaeta* sp. and *Chaetozone setosa* Malmgren, 1867 and the polychaetes *Paramphinome jeffreysii* (McIntosh, 1868) and *Heteromastus filiformis* (Claparède, 1864) (Table II). In addition, nematodes were abundant (detailed species lists are given in Supplementary Material C). All grab samples from Husnesfjorden contained large amounts of spicules derived from dead demosponges from surrounding areas (20–60% of the content of the grab). The average EQR for NQI1 for the Husnes site was 0.95 and the average EQR for H' was 0.90, indicating a high ecological status for this body of water (Table III). The concentration of oxygen in the bottom water was  $5.68 \text{ ml l}^{-1}$ , arriving at the same classification.

## Discussion

### *Chlorophyll-a and nutrient values*

All nutrient and chlorophyll-*a* values in both areas were within the thresholds for high water quality set by the national authorities, and there were no

Table II. Abundance of the five most common infauna species at three deep water sites in Hardangerfjord, Husnes, Varaldsøy and Herand, in 2008–2011.

Husnes	ind. $\text{m}^{-2}$	Varaldsøy	ind. $\text{m}^{-2}$	Herand	ind. $\text{m}^{-2}$
<i>Aphalochaeta</i> sp.	180	<i>Kelliella miliaris</i>	170	<i>P. fiordica</i>	200
<i>Paramphinome jeffreysii</i>	120	<i>C. setosa</i>	150	<i>Thyasira</i> sp.	140
<i>Chaetozone setosa</i>	90	<i>Paradiopatra fiordica</i>	70	<i>Nucula tumidula</i>	30
Nematoda	65	<i>Amphilepis norvegica</i>	70	<i>Terebellides stroemi</i>	25
<i>Heteromastus filiformis</i>	60	<i>H. filiformis</i>	60	<i>H. filiformis</i>	20



Table III. Diversity of the infauna communities sampled in the three deep water sites in Hardangerfjord: Husnes, Varaldsøy and Herand. Richness = total number of species and Diversity = H'; AMBI = AZTI Marine Biotic Index; NQI1 = Norwegian Quality Index 1; EQR = ecological quality ratio. All values are given for a 0.1-m<sup>2</sup> van Veen grab sample.

Station	Sample	Richness	Abundance	Diversity	Mean AMBI	NQI1	EQR NQI1	EQR H'	Status
Herand	1	18	60	3.40	1.88	0.72	0.92	0.77	
Herand	2	15	63	3.06	2.01	0.68	0.88	0.70	
Herand	3	20	48	3.75	2.43	0.70	0.89	0.85	
Herand	4	22	91	3.55	1.68	0.74	0.95	0.81	
Herand	5	20	71	3.34	1.63	0.74	0.95	0.76	
Varaldsøy	1	31	161	4.14	2.20	0.72	0.93	0.94	
Varaldsøy	2	20	38	4.00	2.38	0.71	0.91	0.91	
Varaldsøy	3	28	116	4.22	1.28	0.79	1.01	0.96	
Varaldsøy	4	27	105	4.35	2.14	0.73	0.93	0.99	
Varaldsøy	5	33	158	4.31	1.86	0.75	0.97	0.98	
Husnes	1	39	169	4.62	2.26	0.74	0.95	1.05	
Husnes	2	29	141	4.11	2.09	0.73	0.93	0.93	
Husnes	3	21	82	4.05	1.91	0.72	0.93	0.92	
Husnes	4	19	56	3.80	1.81	0.73	0.94	0.86	
Husnes	5	30	136	3.29	1.57	0.77	0.99	0.75	
Herand	Mean	19	67	3.42	1.92	0.72	0.92	0.78	Good
Varaldsøy	Mean	28	116	4.20	1.97	0.74	0.95	0.96	High
Husnes	Mean	28	117	3.97	1.93	0.74	0.95	0.90	High

indications of elevated levels in the intermediate area, which produces the higher biomass of salmon. The Hardangerfjord is a very dynamic fjord with surface current speed normally in the range of 10–50 cm s<sup>-1</sup> and monthly renewal of the upper fjord water (Asplin et al. 2014). Modelling of the distribution of nutrients in the fjord, taking into account water exchange rates, shows that nutrient emissions from fish farms will increase the natural nitrogen concentrations (Anon. 2011) and phytoplankton biomass by 1–5% (Skogen et al. 2009).

In general the chlorophyll-*a* values were lower in the inner area compared to the intermediate area, indicating a lower primary production in the former (Figure 2). The inner area experienced weak spring blooms compared to the intermediate area, particularly in 2009 and 2010. The spring bloom in 2010 was not very pronounced in any of the areas studied. Braarud (1974a, 1974b) studied the phytoplankton communities in Hardangerfjord during the period 1955–1956 and found an evident fjord effect on the species composition. He found the inner area to have lower biomass of plankton than the outer area during spring and summer, while in autumn the inner part of the fjord could maintain a higher biomass of particularly *Skeletonema costatum* (Greville) Cleve. He also found that the species composition could differ substantially between the inner and outer areas of the fjord. As phytoplankton communities were not studied in detail in this investigation and Braarud's study only lasted one year, it is difficult to draw any conclusions on the reasons for the lower differences between the phytoplankton communities in the inner and outer parts of Hardangerfjord.

Our study does not fully meet the requirements of the WFD for the biological quality element phytoplankton, as we merely measured chlorophyll-*a* values as fluorescence. The WFD requires actual measurements of phytoplankton abundance and studies of plankton composition. However, chlorophyll-*a* values are regarded as a useful expression of phytoplankton biomass and are a responsive indicator of nutrient enrichment in coastal water (Devlin et al. 2007; Harding 1994). The chlorophyll-*a* values measured in Hardangerfjord in this study give no indication of any ongoing eutrophication processes.

#### Macroalgae

The multimetric indices for the macroalgal communities showed high ecological conditions at all sites examined in the intermediate area of Hardangerfjord. In the inner area the macroalgal community at all sites showed a high ecological condition except for the innermost site (31). This site had the lowest species richness of the investigated sites, most likely explained by dense settlement of blue mussels in the intertidal zone or due to salinity stress in the innermost part of the fjord. The physical conditions in Hardangerfjord are known to be highly variable. The entire fjord is influenced by freshwater influx, but with the surface salinity being much lower (< 23) in the innermost fjord arms than in the middle and outer parts of the fjord, where surface salinity is generally >27 due to increasing mixing caused by the effect of winds (Asplin et al. 2014). A high freshwater influence, especially during summer, will exclude several freshwater-intolerant macroalgal spe-

cies from the littoral community. A low intertidal species richness is therefore the rule rather than the exception when moving towards the inner parts of Hardangerfjord (Jorde & Klavestad 1963; Husa et al. 2014). The macroalgal index used in this study is developed for coastal areas and is still not approved for freshwater-influenced fjords; thus, it is not likely that the index is overestimating the ecological condition in the intertidal communities. A number of species on the reduced species list (Anon. 2009) were not present in the inner area of Hardangerfjorden.

Husa et al. (2013) compared macroalgal communities down to 30 m depth in Hardangerfjord with historical data from the 1950s and found that the abundance of habitat building species like kelp and fucoids had a high resilience in the fjord, despite 50 years of anthropogenic activity. They also found that the main drivers of the observed changes in community structure were higher sea temperatures and altered salinity in the fjord due to the hydro-electrical power plant industry. The high ecological condition in macroalgal communities in the intertidal zone is not surprising given the low nutrient level in the fjord. However, we cannot rule out that local impacts on macroalgal communities might occur in the vicinity of farms (< 1 km away). Such impacts have been shown for seagrass meadows and maerlbeds (Hall-Spencer et al. 2006; Diaz-Almela et al. 2008; Duarte et al. 2008; Sanz-Lazaro et al. 2011; Aquado-Giménez & Ruiz-Fernández 2012).

#### *Soft bottom infauna and oxygen*

A relatively species poor and low abundance infauna community dominated by species commonly found in the silty deep basins of western Norwegian fjords, e.g. *Nucula tumidula* Malm, 1861 (Kutti et al. 2007a) and *Paradiopathra fiordica*, was found in the inner and deepest part of Hardangerfjord. The low species richness is in contrast to the general trend of increased infauna species richness with increasing depth (Holte et al. 2004) but in line with early observations of Fauchald (1972, 1974) in that the basins of deep fjords (such as Hardangerfjord and Sognefjord) are often species-poor. Species richness of infauna is known to decrease with decreasing oxygen levels in the bottom water (Buhl-Mortensen et al. 2009). An extreme low infauna diversity (0.99  $H'$ ) and the dominance of the tolerant polychaete *Spiochaetopterus typicus* M. Sars, 1856 were found in this area in 1996 (Rygg & Skei 1997), indicating that periods of oxygen deficiency do occur and that the frequency of bottom water renewal is an important determinant for the ecological status of the deep

benthic communities in the inner part of Hardangerfjord. At the time of our sampling, oxygen levels in the inner part of the fjord were within thresholds of good conditions and only slightly inferior to that of the outer part. The infauna species composition in our study also indicated that there had been no periods of oxygen deficiency in the deep waters of Hardangerfjord during the last couple of years. Infauna abundance is known to increase with both quality and quantity of organic matter arriving at the sea-bed (Dauer & Conner 1980; Rosenberg 1995; Flach & Heip 1996; Kutti et al. 2008). High siltation rates of mineral particles originating from run-off from the large Folgefonna glacier, in combination with the lower primary production in the inner part, could explain the lower total abundance of infauna (i.e. 70 ind.  $m^{-2}$ ) observed in the inner part of the fjord as compared to the outer part (i.e. 120 ind.  $m^{-2}$ ).

The examination targeting the accumulation area at 643 m depth in the intermediate part of the fjord showed no indications of enrichment, neither from the settling out of phytoplankton blooms nor from farm derived particulate waste. The sediment supported a highly diverse infauna community dominated by species characteristic for unaffected areas in western Norwegian fjords, e.g. *Kelliella* sp. and *Amphilepis norvegica* (Kutti et al. 2007a) and *P. fiordica*. The deep basin at 455 m depth, however, supported an infauna community dominated by opportunistic species, i.e. the cirratulids *Aphelochaeta* spp. and *Chaetozone setosa*, *Paramphinoe jeffreysii* and the capitellid *Heteromastus filiformis* (Claparède, 1864). The infauna species composition at this site could be reflecting a beginning of enrichment, caused by the cumulative impact of the production and release of organic waste from several fish farms over many years to the accumulation basin. This is, however, likely to be a slow process, as below the highly dynamic surface layers of the fjord, current speeds are generally low and most waste settles and is processed by macrofauna and microorganisms in a restricted area close to the farms (Kutti et al. 2008). Another explanation for the dominance of opportunists could be the occurrence of large amounts of spicules derived from dead demosponges from surrounding areas. Opportunistic species are known to respond also to other types of disturbances besides organic enrichment, e.g. Bett & Rice (1992) reported a considerable increase in the abundance of macrofauna in sediment samples containing sponge spicule mats relative to samples without them.

## Conclusion

Overall, the benthic and pelagic communities beyond the immediate proximity of fish farms in Hardangerfjord seemed to be little affected by the effluents of nutrients and deposition of organic matter from the salmon farming industry. However, the innermost macroalgal site and the deep bottom fauna in the inner basin were only categorized as in 'good condition' according to the standards of The European Water Framework directive. The high assimilative capacity of nutrient emissions and organic waste is most likely due to the large water volume in the fjord and the dynamic physical environment. The European Water Framework directive requires a precautionary approach, meaning that when several sites are assessed in the same area, the site with the lowest Ecological Quality Ratio score provides the final ecological status for that specific quality element. Similarly, the overall ecological status of the water area should be determined by the quality element giving the lowest score of the quality elements included in the assessment (Anon. 2009). The inner part of Hardangerfjord has a low salmon production, with natural run-off being the main source for nitrogen emissions to the area (Anon. 2011). The fact that the scores of both littoral and deep infauna quality elements resulted in a classification of the ecological status as only 'good' indicates that several other stressors are acting on the communities. While freshwater run-off clearly structured the surface layer and intertidal communities, bottom water renewal may be the main structuring agent for the deep infauna communities.

Although there is little evidence of regional impact from fin-fish farming in Hardangerfjord, the cumulative effect of numerous impacted areas around the fish farms must be taken into consideration when further evaluating the total impact from fin-fish farming on ecosystem functioning in this area.

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