

Ecological Impacts of Emamectin Benzoate, a Sea Lice Therapeutant, on Benthic Communities in the Vicinity of Norwegian Aquaculture Facilities

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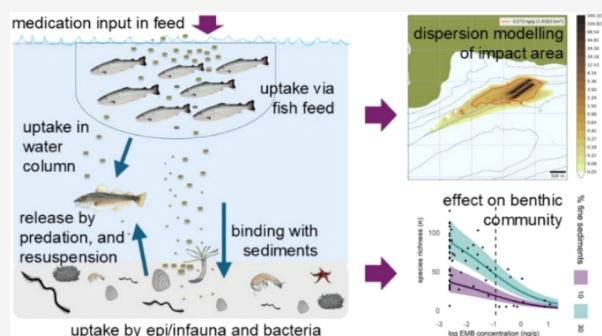
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ABSTRACT: Sea lice (*Lepeophtheirus salmonis* and *Caligus elongatus*) are a major challenge for aquaculture in Norway and beyond. The aquaculture industry uses medicinal (e.g., emamectin benzoate (EMB)) and non-medicinal (e.g., thermolicers) methods to control sea lice. Non-medicinal methods can harm fish, while medicinal treatments can accumulate on the seafloor and impact the environment. Nevertheless, research on EMB's effects on non-target fauna in Norway remains limited. This study is the first to integrate field and modeling analysis of EMB and benthic ecology in Norway, aiming to (i) map EMB distribution around farms, (ii) assess impacts on benthic communities, and (iii) compare EMB's effects on crustaceans to other stressors. Results show EMB had the strongest effect on benthic abundance and species richness, compared to other stressors (e.g., organic matter). EMB concentrations at 0.2 ng/g, near the SEPA EQS of 0.272 ng/g, reduced species diversity and disrupted community structure, with crustaceans being most sensitive. EMB was found to be widespread, persistent, and to accumulate in sediments long after use. Dispersion modeling suggest harmful concentrations could extend 1 km from farms. These findings highlight the need for improved monitoring, regulation, and sustainable lice management to protect biodiversity in aquaculture intensive areas.

KEYWORDS: emamectin benzoate, atlantic salmon aquaculture, sea lice, benthic fauna, benthic crustaceans, environmental impacts, species richness, sensitivity, abundance trends



1. INTRODUCTION

Norway is the world's largest producer of Atlantic salmon (*Salmo salar*) and aims to increase annual production by 2050.¹ To achieve this target, the aquaculture industry must overcome significant challenges, including controlling sea lice infestations, preventing fish from escaping, sourcing sustainable feed ingredients, and eliminating or reducing harmful environmental discharges.

Sea lice (*Lepeophtheirus salmonis*, *Caligus elongatus*) are considered one of the major challenges for the aquaculture industry in fish-producing countries.² Sea lice pose a dual threat by increasing the risk of transmission to wild fish and causing substantial economic losses, both through direct treatment expenses and indirect effects like mortality, reduced growth, health problems, and environmental impacts (e.g.,^{3,4}). Both medicinal (bath and in-feed treatments) and non-medicinal methods are used to control sea lice, but each comes with challenges for fish welfare and the environment (e.g.,^{5,6}).

Emamectin benzoate (herein referred to as EMB), an avermectin, is the active ingredient in the in-feed medication

commercially known as Aquavet SLICE and Slice Vet. EMB has been used in Norway for over two decades, with annual usage ranging from 36 to 259 kg since 2012.^{7,8} In 2022, EMB was the most prescribed medicinal salmon lice treatment in Norway, accounting for about 46% of all prescriptions used all along the Norwegian coast.⁹

EMB is administered to fish via in-feed treated pellets. Uneaten pellets and fish waste, containing EMB residues, settle on the sea floor below and around the fish cages. Studies show that significant amounts of unaltered EMB also enter the water column through fish feces and urine, carrying EMB residues not absorbed by the fish.^{10,11} EMB has distinctive characteristics, including a long half-life in marine sediments, with a persistence ranging from months to years, and is very

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hydrophobic (log Kow of 5.9 at pH 9) (e.g.,^{12–14}). Because EMB persists in sediments around fish farms, it can continue to expose non-target benthic organisms long after treatment—either through direct contact with the sediment or by consuming contaminated material or prey.¹⁵ Extended exposure to elevated concentrations increases the risk of adverse effects on non-target species.

EMB exerts its impact on organisms by interfering with the nervous system and nerve impulses, leading to paralysis in the neuromuscular system and eventual death. This direct effect has been extensively documented, as outlined by studies such as that of Burridge.¹⁶ The sensitivity to EMB varies significantly among non-target marine organisms, with crustaceans emerging as the most susceptible group, a characteristic aligned with EMB's primary function in combating the parasitic crustacean copepod (sea lice). Benthic crustaceans play a vital role in ecosystem functions such as bioturbation, and many benthic species function as ecosystem engineers—modifying sediment structure, driving nutrient cycling, and shaping habitats in ways that enhance biodiversity.¹⁷ Given their importance, it is crucial to understand how EMB impacts these and other non-target organisms. Most studies on EMB have been conducted in controlled laboratory settings focusing on single species; however, field research assessing the effects of EMB on benthic communities remains limited.

EMB residues in sediments around fish farms can exceed critical thresholds, potentially harming marine organisms.^{18,19} To address this, Environmental Quality Standards (EQS) for EMB have been developed with the aim of protecting benthic populations. While Norway lacks a specific EMB EQS, other fish-producing nations, like Scotland, regulated by the Scottish Environmental Protection Agency (SEPA), have established an EMB specific EQS.^{20,21} Scotland's new EQS for EMB (from 2024) has sparked debate, with salmon producers concerned about treatment limitations compared with environmental groups calling for a complete ban. Norwegian research communities often use Scotland's EQS for risk assessments and environmental impact evaluations around farms.

Studies from Canada,^{22–24} Norway,^{25,26} Chile²⁷ and Scotland¹⁷ have reported field levels of EMB above the current Scottish EQS in marine sediment around finfish cages. Few field studies have examined and documented the actual impact of EMB on benthic crustacean communities. Since EMB concentrations have been found above the new EQS level in several countries, including Norway, and negative effects on benthic crustaceans have been documented even within EQS limits,^{17,22} there is a clear need for more field studies linking measured EMB concentrations with benthic ecological data. This is particularly important in Norway, where EMB use is high.⁹

This study is the first in Norway to assess EMB concentrations and benthic ecology simultaneously, using an integrated field-based approach. Residues of EMB and other relevant chemical variables (e.g., nutrients) were analyzed alongside benthic ecological data. In addition, dispersion modeling was conducted. The aim was to (i) analyze and describe the distribution and residue concentrations of EMB in marine sediments surrounding Atlantic salmon aquaculture farms in the Norwegian coastal region, including both a field and oceanographic modeling approach; (ii) evaluate the implications of field residues of EMB on the benthic community; and (iii) specifically investigate the effects of

EMB residues on benthic crustaceans in comparison to other in situ influences such as nutrient levels.

2. MATERIALS AND METHODS

2.1. Field Methodology

Eighteen salmonid marine fish farms along the Norwegian coast were surveyed from 27/07/2021 to 16/11/2022 by Akvaplan-niva and Åkerblå during regular C-surveys (see description below) and associated Aquaculture Stewardship Council (ASC) certifications. Six localities at lower latitudes, between 61° and 63°N, and 12 localities at higher latitudes, between 68° and 71°N, were investigated. The study focused on EMB, although other in-feed treatments (flubenzuron) were also analyzed. However, due to limited samples of both teflubenzuron (TEF) and diflubenzuron (DFB), the emphasis remained on EMB.

In Norway, the Aquaculture Act of 2005 mandates accredited environmental monitoring of the immediate proximity of the seabed at aquaculture sites, ensuring compliance with the seabed impact standard.²⁸ Monitoring is conducted through the C-survey, which integrates benthic surveys and modeling to evaluate environmental quality. It focuses on macrofauna communities, sediment conditions (including particle composition, pH, and redox), hydrography (temperature and salinity), and environmental pollutants. The survey evaluates environmental quality through macrofauna species count, community dominance, and diversity indices, at 3–5 sampling points (stations) during peak aquaculture production at different distances in current direction from the farm. Although C-surveys do not usually cover in-feed medicine residues, this study included analysis of EMB, DFB, and TEF.

During the C surveys, two parallel sediment samples were taken at 2–7 stations along the transect for both chemical and ecological analysis: one at the cage edge zone (20–35 m from cage edge; local zone), one to three points at a distance (35–450 m; intermediate zone) and one outer point (450–560 m; outer edge of intermediate zone). For ASC certification²⁹ reference stations (970–1800 m from the cage edge) were also sampled. Chemical, geological, environmental (TOC, TOM, nitrogen, phosphorus, copper, and zinc), and biological samples were collected at 2 to 7 stations per site. In total, 81 stations were sampled. A CTD cast was taken closest to the cage for each location, but at different times of the year (January to December) depending on the location.

Sampling was performed using a 0.1 m² van Veen grab by field personnel accredited in accordance with NS9410²⁸ and NS-EN ISO 16665.³⁰ At each station, three grab samples were collected: two for macrofauna analysis and one for geological and chemical analysis. In addition to macrofauna and standard parameters for C- and ASC-surveys, residues of in-feed medicines (EMB, TEF, and DFB) were analyzed. Sediment samples were frozen the same day and stored at −18°C until analyzed. Samples for macrofauna analyses were sieved through 1 mm sieves and the remaining material was stained with Rose Bengal and preserved with formalin.

The last treatment date for EMB was obtained from the Norwegian Veterinary Medicinal Products Register “VetReg”³¹ and BarentsWatch,³² which monitors medicine use in aquaculture.

2.2. Laboratory Methodology

In the field, samples were evaluated for sensory parameters and only grabs with an undisturbed surface were approved for sampling (see [Supporting Information \(SI\) 2.2](#) for more information). For chemical analyses, samples were taken from the top 1 cm of the surface, while for geological samples (particle size distribution, TOM and TOC), samples were taken from the top 5 cm (see [SI 2.2](#) for more information).

For chemical parameters (Åkerblå samples), a sample was extracted for the analysis of total organic carbon (TOC), total organic matter (TOM), nitrogen (N), phosphorus (P), copper (Cu), and zinc (Zn) from the same grab where a sample was taken for particle size distribution, all of which were analyzed by the accredited subcontractor Eurofins environmental testing, Norway AS. For Akvaplan-niva chemical samples, a sample was extracted for the analysis of TOM, TOC, nitrogen (N), and copper (Cu). Analytical methods and results of the analysis can be found in [SI 2.2](#).

Particle size analysis was undertaken to determine the fraction of the sample below 63 μm which constitutes fine material (see [SI 2.2](#) for more information).

Chemical analysis of EMB, TEF and DFB were measured via LC-MS/MS using an Acquity UPLC system (Waters) connected to a Xevo TQ-S mass spectrometer (Waters). More details can be found in [SI 2.2](#). The Limit of Detection (LOD) for the method was 0.05 ng/g dry weight (EMB). For calculation purposes and statistics 0.05 ng/g dry weight was used (worst case scenario), even though in reality measurements were likely distributed between 0 and 0.05 ng/g.

For the faunal investigations, benthic organisms from the samples were identified by specialists (taxonomists) preferably to species level or another appropriate taxonomic level.³³ More details can be found in [SI 2.2](#).

The Norwegian environmental quality standards for sediment³⁴ were applied to the measured compounds, where available. The sediment quality is classified into five categories, from 1 ("very good") to 5 ("very poor"). These categories reflect increasing deterioration of sediment, informed by previous laboratory tests, risk assessments, and toxicity data.

2.3. Calculating Ecological Indices

Since most soft-bottom species are sedentary, quantitative analyses of benthic fauna provide a good indication of local environmental impacts.

Statistical methods were used to describe community structure and to compare similarities based on species counts. Following guidelines, multiple indices were calculated from species and abundance data (averaged over two replicates; see [SI 2.3](#)) to assess the environmental condition.

The following statistical methods were employed to describe community structure and to assess similarity between the different communities; 1) Shannon–Wiener diversity index (H') and Hurlbert's diversity index (ES100)—expected number of species per 100 individuals (gives the biodiversity), 2) Indicator Species Index (ISI2012) (unsuitable at low individual/species counts) and Norwegian Sensitivity Index (NSI), Sensitivity index included in NQI1 (AZTI's Marine Biotic Index) (gives the sensitivity), 3) Norwegian Quality Index (NQI1) (encompassing both biodiversity and sensitivity), 4) The Normalized Ecological Quality Ratio (nEQR) for benthic fauna measures ecological health by comparing observed species diversity to reference conditions.

2.4. Oceanographic Dispersion Modeling

To explore areas around fish farms at risk from EMB further from the cages, where sediment samples were not taken, a comparison of modeled predicted environmental concentration (PEC) in sediment to effect threshold levels (Predicted no effect concentration PNEC (SEPA EQS)) was performed ([Figure 3a–b](#) and [Figure 4a–b](#)). To perform the PEC simulations, the hydrodynamic model FVCOM³⁵ was coupled with a sedimentation model formulated in FABM.³⁶ In the applications of the model, the resolution was increased in the vicinity of aquaculture facilities and in areas with complex topography. The sedimentation of feed waste and feces during maximum production, as well as simulation of a hypothetical delousing with EMB was performed for two different Norwegian aquaculture localities. One location with larger production (14 cages with a biomass of 3089000 kg fish) and stronger current, and one with lower production (4 cages with a biomass of 61773 kg fish) and weaker current to illustrate the differences between dispersion at different hydrodynamic and biological conditions ([SI 2.4](#) and [Table B1](#) for more details). The model results show how sediments were dispersed as well as the concentrations of EMB that arise in the sediments immediately after delousing, and over time as the concentrations dilute and the chemical degrades (1 month, 10 months and 20 months after delousing). Note that this was only done for two locations, and the conditions and areas of effect will vary depending on fish biomass and the specific circumstances in different areas. Since the medicine is administered for a rather short period of 7 days, the weather conditions during this time could be quite influential on the dispersion pattern.

2.5. Statistical Methodology

Statistical models, both linear and nonlinear, were used to investigate the decrease of EMB concentration in sediments with distance from the farm and its relationship with the diversity and sensitivity of various benthic groups (phyla), with particular focus on crustaceans, expected to be the phylum most negatively impacted. Generalized Additive Models (GAM) were used when the shape of the relationship was unknown, while Generalized Linear Models (GLMs) were used when a logarithmic relationship could be assumed between the variables. Akaike Information Criterion (AIC) was used for backward model selection, where models with $\Delta\text{AIC} \leq 2$ were considered comparable and the simplest model was retained. For models with $2 < \Delta\text{AIC} \leq 10$, significance of single terms was evaluated to assess whether to retain the term ($p \leq 0.01$) or not ($p > 0.01$).³⁷

Depth was very different between latitudinal groups, with an average depth of 105 m (50% of data within 82–118 m range) for higher latitude localities (range 68° and 71°N) and 247 m (50% of data within 128–333 m range) for lower latitude localities, hence either depth or latitude was included in the models. Temperatures showed distinct differences depending on latitude, with higher average temperatures (8 °C) and lower seasonal variation at lower latitudes, and much larger seasonal variation and lower average temperatures (6.5 °C) at higher latitudes ([Appendix \(A\) Figure A5](#)). Not all localities included samples from all the different zones (Cage edge, Transition inner and outer, Outer, Reference, see [Table A1](#)). All samples included measurements of the main parameters (distance from cage, EMB, TEF, DFB depth, latitude, and months since last treatment), but some missing values were present for nTOC, % fine sediments, Cu, Zn, and P (see [Table A1](#)). Cu, Zn and P

measurements were also lacking more often from the higher latitude localities and were therefore not used in the models. Seafloor temperature was measured during different seasons across localities and only at the cage edge and were therefore not included in the models, but the two latitudinal groups represented the general climatological conditions of the stations. For EMB, TEF, and DFB, 51%, 74%, and 91% of the sediment measurements were at or below the detection limit (LOD, 0.05 ng/g for EMB; 0.1 ng/g for TEF and DFB). DFB was detected at measurable concentrations (range 0.5–17.1 ng/g) only when EMB was above the SEPA threshold (0.272 ng/g) suggesting a degree of positive correlation between the two. This was not the case for EMB together with TEF.

2.5.1. Footprint of the Farm: EMB Concentration Decrease with Distance. The EMB concentration was modeled as a function of distance from the farm, time (months) since last treatment, nTOC, % fine sediments, and either bottom depth or latitudinal group (as low or high latitude). Due to some missing values in the predictors (see Appendix Table A1) the model was run on a slightly reduced data set ($n = 65$). The model used was a GLM with log-link and a Gamma distribution to account for overdispersion in the residuals. Logarithmic transformations in the predictors were evaluated for highest deviance explained and linearity of the relationship.

2.5.2. EMB Impacts on Species' Richness, Sensitivity, and Abundance. GAMs were used to investigate the shape of the relationship between species diversity and EMB concentration, accounting for possible nonlinear patterns and for the presence of potential threshold distances of negative impact. This was done for all taxa pooled together as well as separately for selected phyla (AN = Annelida, CR = Crustacea, EC = Echinodermata, MO = Mollusca) to outline possible differences in responses among different taxonomical groups. The same approach was used to investigate patterns of overall sensitivity (independent of taxon) with increasing EMB concentrations.

GAMs with a Negative Binomial distribution and log link (accounting for overdispersion) were used to predict species richness using a set of available predictors: EMB, TEF, DFB, nTOC, depth, % fine sediments and latitudinal group. Due to the absence of measurements for nTOC and % fine sediments in one of the reference stations (distance >600 m), sample size was reduced to $n = 64$. Separate models were fitted for all taxa and for overall species richness across taxa. A logarithmic transformation was used for EMB and TEF in all models. Due to the low number of measurements above detection level, DFB was included as a presence/absence factor variable in addition to EMB. Due to its partial correlation with EMB, which could create a confounding effect, an additional categorical variable was tested with 3 classes: "no chemicals" (both EMB and DFB under detection level), "EMB only" (only EMB above the detection level) and "EMB and DFB" (both above detection level). Note that DFB was not detected in any sample alone (i.e., without EMB being detected). Depth and latitudinal groups (included as a fixed factor variable with two levels, high and low latitudes) were included in separate models to evaluate predictor performance.

The same approach was used to predict species sensitivity (Norwegian Quality Index—NQI, $n = 63$), using a Beta distribution and logit link, as the values are bound between 0 and 1, and abundances ($n = 62$) using a negative binomial

distribution and log link to account for overdispersion. When modeling abundances, species sensitivity (NQI) was also included as a predictor.

3. RESULTS

3.1. Survey Results

A total of 81 sediment samples taken from varying distances from the cage edge at 18 different aquaculture localities were included in the analysis (for calculations of means at different zones). Some of these data points did not include all the chemical parameters and some were not suitable for calculation of all the faunal indices (unsuitable at low individual/species counts). An overview of data points included for each parameter is presented in Tables 2 and 4. For prediction of species richness, based on a set of available predictors, a total sample size of 63 was used (see SI 2.5).

3.1.1. Chemistry Results. **3.1.1.1. In-Feed Medicinal Residues in the Environment. Results from Sampling:** All the farms had registered use of EMB. 58% of sediment samples collected for analyses of in-feed medical residues had concentrations of EMB above the limit of detection (LOD 0.05 ng/g dry weight (DW)). EMB was detected at all farms surveyed, except two. Not unexpectedly, these two farms had the longest periods since the use of EMB, with durations of 35 and 70 months since the last treatment. Concentrations generally followed a spatial gradient linked to distance from the cages, with the highest concentrations found in the immediate vicinity of the cages (Figure 1, mean concentrations Table 2).

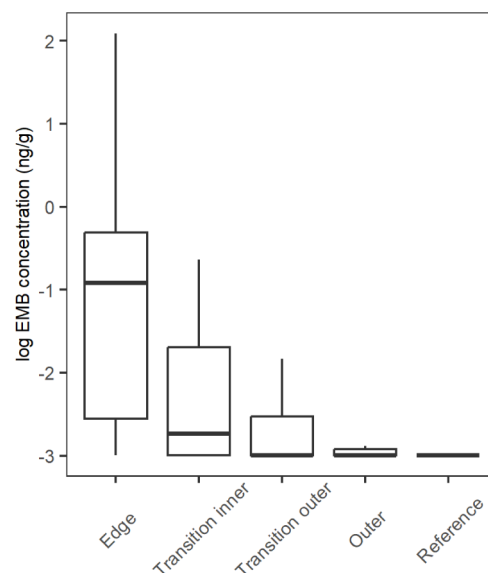


Figure 1. Box plots of log EMB concentration in sediment by sampling zone (cage edge = close to the edge of the fish farm 20–35 m; transition inner = 50–200 m from cage edge; transition outer = 201–450 m from cage edge; outer = 451–560 m from cage edge; reference = reference stations >970–1815 m from cage edge).

Approximately 21% of the samples exceeded the 2023 SEPA EQS for sediment EMB (0.272 ng/g DW). About half of these samples were found near the fish farm (20–35 m), with 25% located in the inner and 25% in the outer transition zones. Additionally, the farms associated with these samples used EMB between 5 and 22 months prior to sampling.

The full GLM model explained deviance and linearity in the log-link domain better when using a log transformation for both distance to the farm and months since last treatment (Δ deviance explained 13%). The full model with latitudinal group had a slightly higher deviance explained (6%) than the one with depth, so the first was kept as a predictor in model selection. AIC model selection resulted in log distance, log months since treatment and latitudinal group being kept in the model. As the predictors with missing values were removed during model selection, the final model was refit with all available samples ($n = 81$) and deviance explained remained virtually unchanged (Δ deviance explained -0.3%) despite a 25% increase in sample size.

Months since last treatment and distance showed the negative effects on EMB levels (Table 1), while latitudinal

Table 1. Summary Statistics of the GLM Model for EMB Including Coefficients Estimates and Standard Errors

Term	Estimate	St. Error	Z value	Pr ($> z $)
Intercept	0.8368	0.6143	6.246	$2.14e^{-08}$
Log (distance)	-0.5260	0.0726	-7.248	$2.81e^{-10}$
Log (months since treatment)	-1.4004	0.1717	-8.156	$5.09e^{-12}$
Latitude group (high lat)	0.8584	0.1782	4.817	$7.15e^{-06}$

group showed a positive effect, with concentrations of EMB being generally higher at higher latitudes at similar distances, depths, and months since last treatment. The decrease of EMB with distance and months since last treatment was linear on a log–log scale, meaning that in absolute values the decrease was sharp at short distances (Figure 2) and a few months after

treatment. Figure 2 shows how the footprint of the farm is smaller in space (distance from farm) at lower latitudes and when longer time has passed since last treatment.

Results from Oceanographic Dispersion Modeling: Modeled PEC concentrations of EMB showed that it generally followed a spatial gradient linked to distance from the cages, with the highest concentrations found in the immediate vicinity of the cages (Figure 3a–b and Figure 4a–b, Appendix A1–A4) in the main direction of the current. Figure 3 a–b and Figure 4a–b show simulated EMB concentrations 30 days after the delousing started. Figures A1a–b and A4a–b show simulated concentrations 10 and 20 months after delousing. SEPA EQS values were used as the PNEC to map areas above and below threshold levels. The results indicate that these thresholds are likely to be exceeded for both sites. After 30 days, hazardous concentrations are expected and may extend over approximately 1 km from the farm (Figures 3a and 4a). After 10 months, hazardous levels can still be detected up to 500 m away (Figure A1a). At site b, the area with hazardous concentrations were smaller and extended over a shorter distance compared to site a (Figures 3b, 4b and A1b).

Flubenzurons: Two of the farms had registered the use of the active ingredient diflubenzuron and all the sampling zones around these farms showed detectable residues in the sediments (Table 2). The measured levels exceeded the Norwegian environmental quality standards²⁶ for diflubenzuron (NEQS sediment; 0.2 ng/g DW) as far as 430 m from the cage edge, indicating that these compounds may pose a risk to benthic marine species living near these fish farms. No samples in the outer and reference zones were taken during the surveys. Based on the results of this study, the sample group “Edge” was

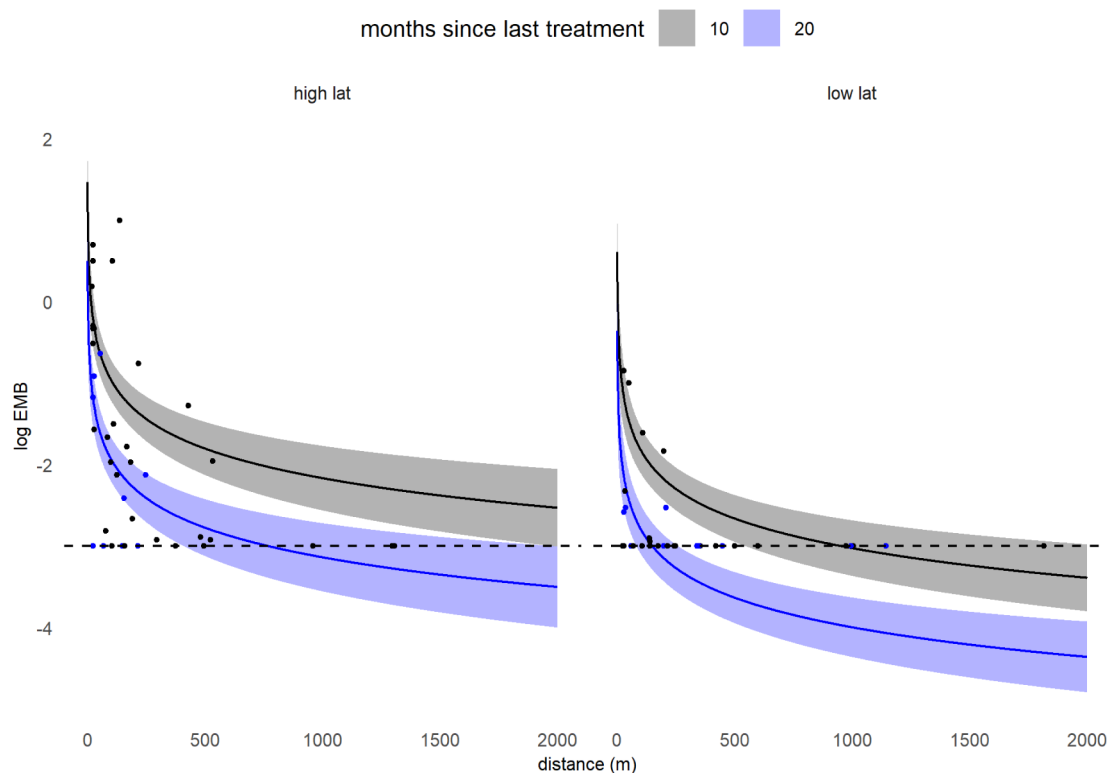


Figure 2. Predicted log EMB decreases with distance from farm (lines) for high and low latitudinal groups (panels) after 10 and 20 months since the last treatment (colors). Measurements are shown as dots. The dashed lines show the lower sensitivity limit of the EMB detection analysis ($\log(0.05 \text{ ng/g}) = -3$). The decrease in $\log(\text{EMB})$ is linear with $\log(\text{distance})$ but is shown in this figure against nontransformed distance.

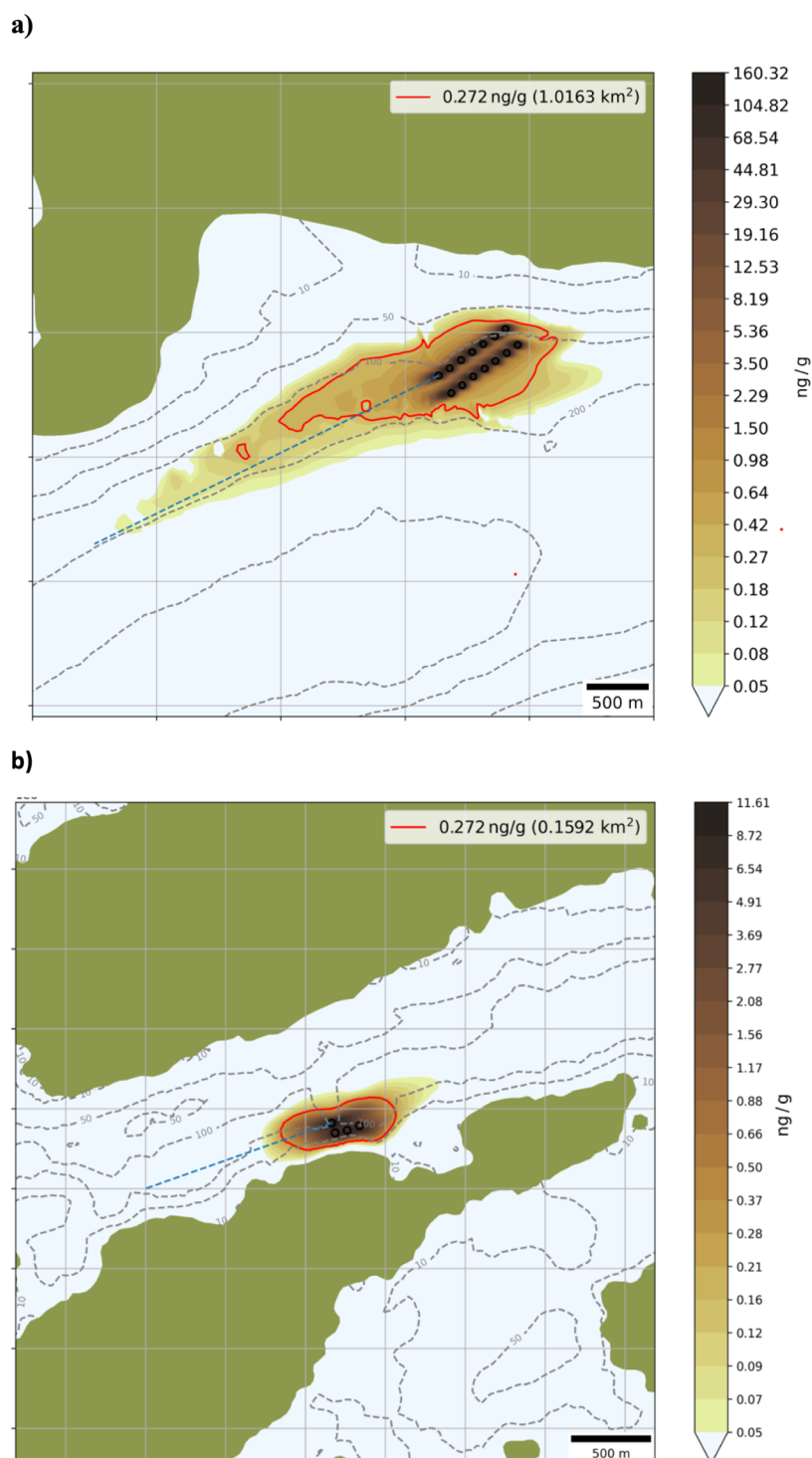


Figure 3. Dispersal model results showing the concentrations of EMB in sediments after simulated delousing operation of 14 cages 1 month since the last treatment. (a) dispersal at high biomass and water current site; (b) dispersal at lower biomass and water current site. Contours of 0.272 ng/g (red solid line) indicate areas with levels of EMB over the SEPA EQS. The blue dashed line is the direction of the strongest current from the aquaculture site, and EMB concentrations along this line is given in Figure 4.

classified as Category IV (Bad), while the “Transition Zone” was classified as Category III (Moderate) according to the NEQS standards.

Only one farm had registered use (prescribed in VetReg but not registered in Barents Watch) of the active ingredient teflubenzuron 6 years ago, and samples from this farm showed high levels in the sediment. These were all above NEQS²⁸ for teflubenzuron (NEQS sediment; 0.0004 ng/g DW), and all

stations were classified as 5—“very bad”, also 1800 m away from the pens. Teflubenzuron was also discovered in sediment from 7 other farms where no previous use was registered. The concentrations at these localities varied from 0.145–5 ng/g DW, ranging from 4—“bad” to 5—“very bad”, although the limit of quantification in this study was 0.1 ng/g DW meaning that other classes (moderate and good) could not be detected.

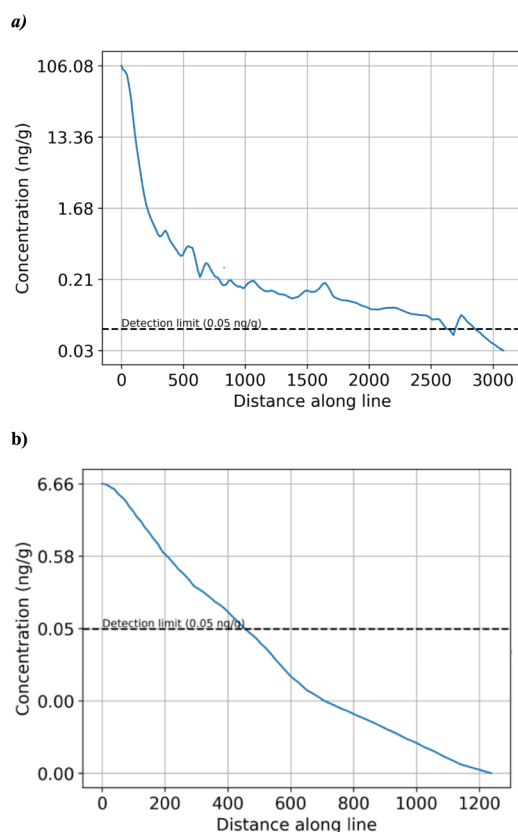


Figure 4. Dispersal model results showing the concentrations of EMB in sediments with distance from farm, following the blue dashed line in Figure 3, after the simulated delousing operation of 14 cages 1 month since the last treatment; (a) dispersal at high biomass and water current site; (b) dispersal at lower biomass and water current site. The dashed lines show the lower sensitivity limit of the EMB detection analysis (0.05 ng/g).

3.1.1.2. Other Measured Chemical Parameters. Mean copper (Cu) concentrations at all sampling stations were classified as good or very good. However, sediment samples at some stations were classified as condition 4 “bad” (see Table 2). Furthermore, 18% of the samples were above the ASC²⁹ environmental limit of 34 mg/kg DW and these concentrations were found in both the edge, transition, and outer zones.

Mean concentrations of zinc (Zn) at all sampling stations were classified as “very good” (Table 2), however, for this parameter some sediment samples showed concentrations that were classified as “good”, or the highest concentration was classified as “moderate” condition.

Also, for normalized total organic carbon (TOC) there are environmental quality standards, and all the inner stations were classified as “moderate” condition, while the outer stations were classified as “good”.

Total nitrogen (TN) and phosphate (TP) do not have environmental quality standards in Norway. However, mean TN was not different between stations, while mean TP was highest near the cages (Table 2). Means for TN and TP at different distances from the cages are given in Table 2, as well as the mean fraction for particles in the sediment <63 μm . The predominant variability in particle size was observed among different sites.

3.1.2. Soft Bottom Fauna, Fauna Indices, and Ecological Condition. Results from C Surveys. Macrofaunal

analysis revealed significant variations in both the abundance of individuals and taxa across different stations and distances from the cages. A total of 19 phylum, 20 classes and 685 taxa were identified across all sediment samples (Table 3). Polychaeta worms emerged as the most diverse and abundant group in the study, followed by mollusks and crustaceans (Table 3).

The findings underscored a discernible gradient in the community impact response relative to the distance from the cages, with the most pronounced impact observed directly in the vicinity of the cages (20–35 m). Close to the cages, the number of individuals was highest, except for crustaceans (Table 4). The pattern reversed when examining species diversity, revealing that the number of taxa was greater at a distance from the cages (Table 4). This also holds true for crustaceans.

Three biodiversity indices were considered (Table 4): species richness (overall and per phylum), Shannon–Wiener diversity (H'), and Hurlbert’s diversity (ES100). These indices were lowest nearest to the cages (20–35 m), where sediments were classified as “poor” to “moderate” according to the NEQS. Further away, the indices increased. Similar trends were observed for the sensitivity and biodiversity indices, including the Indicator Species Index (ISI2012), Norwegian Sensitivity Index (NSI), and AMBI. The outer stations were classified as “good” or “very good” for all indices. The mean normalized ecological quality ratio (nEQR) for stations that were above the SEPA eqs (0.272 ng/g) concentration were 0.35 (stdev 0.20), quantified as “poor” quality by the NEQS.

Species richness, sensitivity and abundance trends. The selected GAM models for species richness included log EMB and % fine sediments for all phyla pooled (ALL), Crustacea (CR), and Mollusca (MO, Table 5). For Annelida (AN), log EMB, % fine sediments and nTOC were significant, for Echinodermata (EC), log EMB, % fine sediments and log TFB were significant, the latter with an unexpected positive relationship with species richness. A sensitivity analysis showed that this result was entirely driven by a high species richness in echinodermata at one site with high TFB concentrations. Upon exclusion of this site, the best model included log EMB and % fine sediments, in line with the results from the other taxa (Table 5, EC*). Due to this, TFB was excluded from the best explanatory model for EC. Deviance explained was highest for AN and lowest for EC.

The results showed overall negative relationships between species richness (log transformed due to the log link) and EMB concentrations (log transformed predictor in the models) for all taxa, confirming the assumed general negative effect of EMB on species diversity (Figure 5). All taxa, except for CR, showed the steepest decrease in species richness around the SEPA threshold level of EMB concentration of 0.272 ng/g DW and presented EMB smoothers with effective degrees of freedom (edf, a measure of the “wiggleness” of the curve) above 2 (range 2.2–2.6, Table 5). CR, on the other hand, showed a sharp drop in species richness already at low EMB concentrations, with a smoothed curve with edfs = 1.5 (edf = 1 is a line), which can be approximated to a linear decrease in logarithmic space (i.e., a logarithmic decrease).

The shape of the relationship between species richness and % fine sediments was often represented by smoothers showing steep increasing richness up to roughly 30–40% of fine sediments, then decreasing values after that, except for CR for

Table 2. Mean, Range, and Stdev of Chemistry Parameters for the Whole Survey by Distance Zone (Cage Edge = Close to the Edge of the Fish Farm 20–35 m; Transition Inner = 50–200 m from Cage Edge; Transition Outer = 201–450 m from Cage Edged; Outer = 451–560 m from Cage Edge; Reference = Reference Stations >970–1815 m from Cage Edge)^{abc}

Chemical parameter		Edge	Transition inner	Transition outer	Outer	Reference	Samples included	Farms included	EQS
EMB (ng/g DW)	Mean	0.92	0.28	0.1	0.06	0.05	81	18	0.272 ng/g DW *
	Range	<LOD - 8.08	<LOD - 2.72	<LOD - 0.47	>LOD - 0.14	>LOD - 0.05	81	18	
	Stdev	1.82	0.59	0.11	0.03	<LOD	81	18	
DFB (ng/g DW)	Mean	6.80	4.30	3.81	-	-	6	2	0.2 ng/g DW**
	Range	(1.70 - 17.10)	(0.50 - 8.10)	-	-	-	6	2	
	Stdev	8.92	5.38	0.00	-	-	6	2	
TEF (ng/g DW)	Mean	802.9	210.1	21.3	-	55.2	6	1	0.0004 ng/g DW***
	Range	-	-	15.9 – 26.6	-	28.5 – 81.7	6	1	
	Stdev	-	-	7.6	-	37.7	6	1	
Cu (Copper mg/kg DW)	Mean	30.25	16.60	21.45	21.20	24.79	67	18	>20 mg/kg DW**
	Range	(7.20 - 99.9)	(5.27 - 41.00)	(5.90 - 39.10)	(5.00 - 48.00)	(9.05 - 33.2)	67	18	>34 mg/kgDW***
	Stdev	24.00	10.10	10.45	13.74	10.73	67	18	
Zn (Zinc mg/kg DW)	Mean	66.85	53.60	55.87	55.11	69.53	54	10	>90 mg/kg DW**
	Range	(16.6 - 143.00)	(18.7 - 208.00)	(20.5 - 88.4)	(14.6 - 106)	(42.7 - 93.2)	54	10	
	Stdev	43.22	44.24	32.47	31.35	25.24	54	10	
TOM (%)	Mean	5.51	3.82	5.74	7.86	-	67	18	
	Range	(1.3 - 19.8)	(1.3 - 9.13)	(0.8 - 14.4)	(2 - 19.9)	-	67	18	
	Stdev	4.88	2.41	1.86	6.27	-	67	18	
nTOC (mg/g)	Mean	27.60	22.80	23.32	23.17	23.75	73	16	<20 mg/g**
	Range	(18.4 - 72.5)	(16.0 - 33.0)	(15.7 - 33.6)	(16.8 - 36.9)	(22.2 - 23.3)	73	16	
	Stdev	13.53	4.44	4.45	3.12	2.19	73	16	
TN (mg/g DW)	Mean	2406.25	2076	2673.33	2727.27	-	70	18	
	Range	(500- 5700)	(400-5100)	(500 - 6500)	(1200 - 6800)	-	70	18	
	Stdev	1650.24	1242.07	2000.43	2049.92	-	70	18	
TP (mg/kg DW)	Mean	2423	1549.15	1189.11	1122.26	1006.75	52	10	
	Range	(735 - 6540)	(510 - 4790)	(634 - 1590)	(756 - 1610)	(994 - 1060)	52	10	
	Stdev	1688.27	1084.96	618.85	565.27	502.58	52	10	
% fine sediment	Mean	38.49	44.11	55.52	47.26	-	65	18	
	Range	(9.9 - 76.9)	(3.9 - 84.3)	(18.3 - 94)	(16 - 87.7)	-	65	18	
	Stdev	22.78	21.73	20.30	23.34	-	65	18	

^a* The SEPA-derived EQS for sediment (SEPA 2024) ^b** The Norwegian Environmental Quality Standards NEQS (Norwegian Environment Agency 2023) ^c***ASC (Aquaculture Stewardship Council) Environmental Limits for Copper (ASC 2023). Bold letters indicate values above standards, and the Norwegian Directorate Group's Guidance was used to classify with five classes: I. Background (Blue), II. Good (Green), III. Moderate (Yellow), IV. Poor (Orange), and V. Very Poor (Red), where Available.

Table 3. Taxonomic Breakdown of Dataset of All Species Recorded in the Sediment Samples (The Benthic Organisms Were Identified by Specialists (Taxonomists) Primarily to the Species Level or Another Appropriate Taxonomic Level

Phylum	Number of taxa (or appropriate level)
Annelida	338
Mollusca	151
Arthropoda	115
Echinodermata	44
Cnidaria	9
Others	28
Total	685

which the relationship was linear and negative in the log domain (Appendix Figure A6).

The full models for all taxa were also tested by exchanging depth with latitudinal group (as a 2-levels factor). Deviance explained was very similar across phyla (mean Δ % deviance explained across taxa was 0.5% higher using latitudinal group), however EMB concentrations were very different between latitudinal groups, so depth was preferred as a predictor.

To investigate species sensitivity to EMB, the ecological index NQI was used, which weights species according to their sensitivity to different types of pollution. A sediment body dominated by sensitive species yields a high NQI and indicates good ecological status, whereas dominance by tolerant species (e.g., certain polychaetes) results in a low NQI and poorer status. NQI scores were pooled across all phyla, the selected model included log EMB, log TFB and depth in linear forms, and a smoother for % fine sediments in the logit domain. This

model showed the same problem than the one for EC species richness, where a significant positive relationship with log TFB was entirely driven by values of high sensitivity at a site with high TFB concentration. Removing the site showed no significant relationship with log TFB, hence this term was not included in the final model (see Table 6 for model summary). The relationship between sensitivity and log EMB was best represented as a decreasing logistic curve (i.e., due to the logit transformation, Figure 6).

For number of individuals (abundance) for all phyla pooled, the selected model included depth and nTOC, but not EMB, suggesting that EMB does not explain the increased number of species that is seen closer to the cages (Table 4). This trend with distance, in fact, exists also at low EMB concentrations (Figure 7). Abundances showed a nonlinear decrease for increasing values of depth and nTOC (Appendix Figure A7). Sensitivity was not significant, indicating that high abundances are not driven by low sensitive species. The overall deviance explained, however, was low (27%), suggesting a lack of strong predictor for abundances among the available measurements (see Appendix Table A2 for the model's summary statistics).

For crustacean abundance, the selected model included only log EMB (edf = 1.6, ~linear in the log domain), but only had a total deviance explained by 31%. For crustacea, log abundance was highly correlated with log species richness (cor = 0.82), which explains the similarity with the results on CR species richness, for which EMB is a significant predictor, while this was not true for overall abundance (cor = 0.35, Appendix Figure A8).

Table 4. Mean, Range, and Stdev of Ecology Parameters for the Whole Survey by Distance Zone (Cage Edge = Close to the Edge of the Fish Farm 20–35 m, Transition Inner = 50–200 m from Cage Edge, Transition Outer = 201–450 m from Cage Edged, Outer = 451–560 m from Cage Edge, Reference = Reference Stations >970–1815 m from Cage Edge)^a

Ecological parameter		Edge	Transition inner	Transition outer	Outer	Reference	Samples included	Farms included	Limits
Number individuals	Mean	1222	931	914	578	501	81	18	
	Range	(4 - 2274)	(52 - 2642)	(92 - 5993)	(127 - 2199)	(61 - 1162)	81	18	
	Stdev	726	662	1316	519	461	81	18	
Species richness	Mean	30	64	65	69	71	80	18	
	Range	(5 - 128)	(6 - 143)	(7 - 99)	(31 - 108)	(16 - 136)	80	18	
	Stdev	33	37	27	28	40	80	18	
Number individuals (Crustaceans)	Mean	6	15	18	12	34	80	18	
	Range	(0 - 46)	(0 - 46)	(1 - 109)	(0 - 39)	(0 - 194)	80	18	
	Stdev	11	14	32	14	62	80	18	
Number species (Crustaceans)	Mean	2	5	4	6	5	80	18	
	Range	(0 - 12)	(0 - 18)	(1 - 6)	(0 - 15)	(0 - 19)	80	18	
	Stdev	3	6	4	10	6	80	18	
NQI	Mean	0.41	0.65	0.71	0.79	0.78	78	18	0.72*
	Range	(0.19 - 0.75)	(0.23 - 0.81)	(0.23 - 0.78)	(0.75 - 0.82)	(0.73 - 0.86)	78	18	
	Stdev	0.17	0.17	0.13	0.02	0.04	78	18	
H	Mean	1.58	3.66	3.95	4.48	4.42	79	18	3.2*
	Range	(0.16 - 4.85)	(0.32 - 5.37)	(0.32 - 5.15)	(3.38 - 5.61)	(2.42 - 5.45)	79	18	
	Stdev	1.52	1.34	1.19	0.47	0.87	79	18	
ES100	Mean	8.93	24.63	29.66	35.00	36.43	73	18	19*
	Range	(0.66 - 13.77)	(1.14 - 42.21)	(15.10 - 39.40)	(29.77 - 41.18)	(30.25 - 47.28)	73	18	
	Stdev	10.20	11.02	7.40	4.18	5.73	73	18	
ISI2012	Mean	6.12	8.23	8.92	9.52	9.50	77	18	8.7*
	Range	(3.57 - 9.39)	(4.02 - 10.28)	(4.02 - 10.28)	(8.36 - 10.50)	(8.20 - 10.26)	77	18	
	Stdev	1.78	1.45	1.34	0.69	0.63	77	18	
NSI	Mean	12.30	19.00	21.11	23.62	24.41	77	18	25*
	Range	(7.04 - 22.62)	(8.7 - 24.53)	(6.95 - 24.94)	(20.73 - 25.55)	(21.61 - 26.71)	77	18	
	Stdev	5.25	4.53	4.23	1.61	1.61	77	18	
nEQR	Mean	0.35	0.69	0.75	0.85	0.84	79	18	0.8*
	Range	(0.14 - 0.84)	(0.13 - 0.90)	(0.13 - 0.89)	(0.82 - 0.87)	(0.75 - 0.92)	79	18	
	Stdev	0.24	0.21	0.17	0.02	0.05	79	18	

^a* Norwegian Directorate Group's Guidance was used to classify with five classes: I. Background (Blue), II. Good (Green), III. Moderate (Yellow), IV. Poor (Orange), and V. Very Poor (Red), where available

Table 5. Selected Models by Taxon in Decreasing Order by Total Deviance Explained^a

Taxon	Dev. Expl.	Term	Edf	Ref df	Chi sq	P-value
AN	66.3%	s(logEMB)	2.235	2.714	54.88	<2e ⁻¹⁶
		s(sediment)	3.448	4.336	20.90	0.000544
		s(nTOC)	4.155	5.094	16.50	0.006175
MO	65.6%	s(logEMB)	2.271	2.817	66.61	<2e ⁻¹⁶
		s(sediment)	4.693	5.768	27.85	0.000154
ALL	61.2%	s(logEMB)	2.182	2.707	79.98	<2e ⁻¹⁶
		s(sediment)	3.761	4.672	21.78	0.000467
CR	46.9%	s(logEMB)	1.475	1.784	36.771	<2e ⁻¹⁶
		s(sediment)	1	1	9.691	0.00185
EC	47.6%	s(logEMB)	2.594	3.221	20.459	0.00022
		s(sediment)	3.045	3.803	8.994	0.04732
EC*	(60.8% ^b)	s(logEMB) ^b	2.572 ^b	3.191 ^b	24.35 ^b	5.37e ^{-05b}
		s(sediment) ^b	3.307 ^b	4.117 ^b	19.54 ^b	0.000759 ^b

^aThe terms included in the selected model are shown with the Respective Effective Degrees of Freedom (Edfs, >1 = nonlinear) and Chi-square statistics (reference degrees of freedom, Chi-square, and probability value). The additional model for EC, excluding one station with high species richness and high TFB levels, is also reported as EC*. Note that deviance explained for EC* is not comparable to the others as it is based on a reduced dataset, hence in brackets ^bSummary statistics for an additional model excluding one station with high species richness of EC and high TFB levels.

4. DISCUSSION

4.1. Sea Lice Medicines in the Environment—Concentrations of EMB and Flubenzurons Decrease with Distance

EMB released into the water column during medicinal in-feed Aquavet SLICE and Slice Vet treatment dissipates quickly, whereas most of the EMB reaching the sediment remains within 150 m of the cage. Studies from Norway, Scotland, Chile, and Canada (Table B2 and references therein) show that the highest EMB concentrations are usually found near the

pens, and lower levels have been detected up to 500 m away, in concordance with this study.

However, results from this field investigation also showed that EMB concentration, measured in sediment samples taken 200–500 m from the farms, was still relatively high, close to or above the threshold levels for environmental impacts set by the Scottish authorities.²¹ Approximately 25% of the samples that were above EQS were found between 200 and 500 m away from the farms. This aligns with a recent Norwegian study²⁶ that measured EMB spread from two farms using different feeding methods. EMB was found in water, marine snow, and

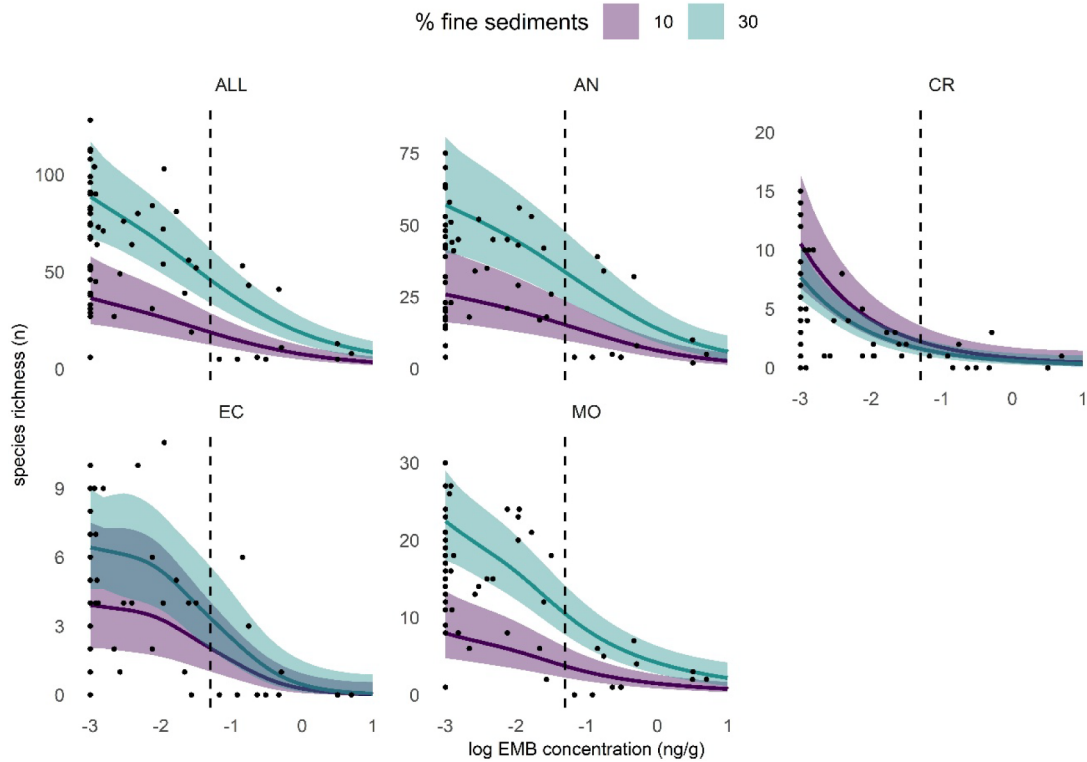


Figure 5. Species richness patterns by phylum (ALL = all phyla pooled, AN = Annelida, CR = Crustacea, EC = Echinodermata, MO = Mollusca) predicted by GAM models (line = mean, shade = 95% confidence intervals) and measured (dots) for increasing EMB concentrations and for different % of fine sediments (colors). Vertical dashed lines show the logarithm of the SEPA threshold value in EMB concentration of 0.272 ng/g = -1.3 .

Table 6. Summary Statistics of the Model for Species Sensitivity (NQI) for All Phyla Pooled Including Coefficient Estimates and Standard Errors for the Linear Terms, and Effective Degrees of Freedom and Chi Square Statistics of the Smoother for % of Fine Sediments

Term	Estimate	St. Error	Z value	Pr (> z)
Intercept	−0.4091	0.1433	−2.855	0.0043
LogEMB	−0.4527	0.0589	−7.688	1.49×10^{-14}
	edf	ref df	Chi sq	p-value
s(sediment)	3.554	4.422	10.52	0.0452

sediments before, during, and after treatments, with residues near or above threshold levels (0.1–1.2 ng/g DW) still detected 500 m away four months after delousing.

In the current study, at distances of 900–1800 m from the pens, samples were below the limit of quantification. This can be because the few reference samples in the ASC survey were taken outside areas influenced by aquaculture and not necessarily downstream of the pens, so they do not accurately reflect EMB concentrations further along with the current flow.

The oceanographic modeling results from a simulation of in-feed delousing at two different aquaculture locations also indicates that higher concentrations are found near the cages, with lower concentrations predicted further away. The affected area (values exceeding SEPA EQS) varied depending on the ecotoxicological data used (in this case, SEPA values and an EMB half-life of 230 days) as well as time. Nevertheless, the modeling results suggest that exceedances probably will occur. After 30 days, hazardous concentrations are expected, and the modeling shows that these concentrations can be detected

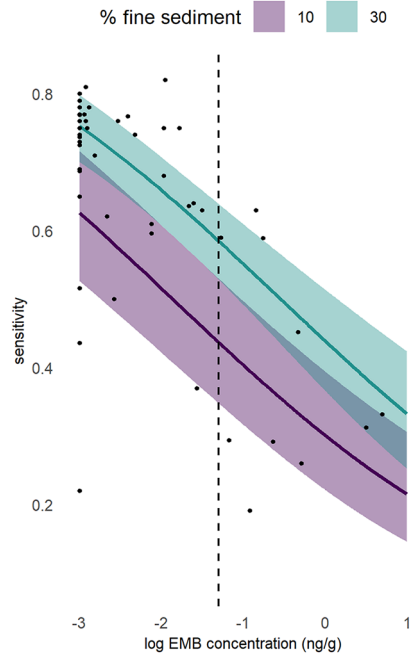


Figure 6. Species sensitivity (NQI) patterns for all phyla pooled predicted by the GAM model (line = mean, shade = 95% confidence intervals) and measured (dots) for increasing EMB concentrations and for different % of fine sediments (colors). The vertical dashed line shows the logarithm of the SEPA threshold value in EMB concentration of 0.272 ng/g = -1.3 .

approximately 1 km from the pens, and more than 500 m away after 10 months at the site with higher fish biomass.

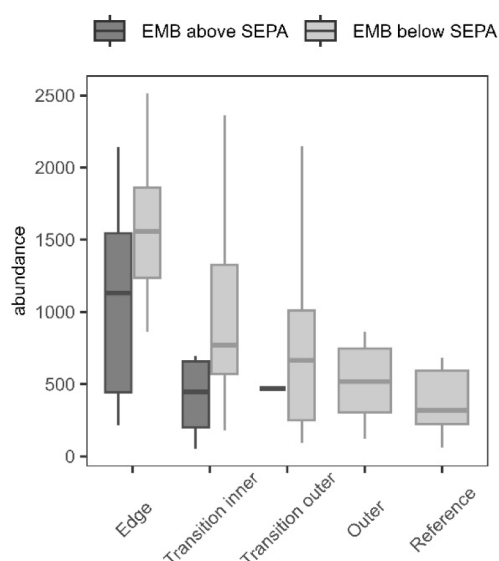


Figure 7. Number of individuals (abundance) independent of taxon by distance zone (*x* axis) for samples with EMB concentrations above and below SEPA threshold levels (0.272 ng/g, respectively dark and light gray box plots).

EMB was detected at 16 of the 18 farms surveyed. The two farms where it was not detected had as expected the longest intervals since EMB was used (35 and 70 months (2.9 and 5.8 years) respectively). However, residue concentrations exceeding the SEPA EQS threshold were still found at one site 22 months (660 days, 1.8 years) after treatment. Similar results have been reported, for example, Sagerup et al.²⁶ found that residue EMB concentrations above the threshold levels for environmental impacts set by SEPA could be detected in samples even though it had been at least 1.5 years (18 months, approximately 540 days) since EMB was last used. Furthermore, a Canadian study reported low concentrations of EMB (0–0.058 ng/g dry weight) from hard bottom flocculent material within 150 m of a facility on the south coast of Newfoundland even two years after it had been followed.³⁸ Another Canadian study documented that EMB was present at low ppb (1 $\mu\text{g/kg}$) at a site over 1.5 years after SLICE treatment.³⁹ EMB was furthermore detected in another Canadian study four years after the last known treatment.²³ This current study, along with others, shows that EMB can persist in sediments beneath and around fish farms, posing a potentially significant long-term risk of exposure to non-target benthic organisms.

There has been considerable debate on the breakdown of EMB in the sediment compartment. The EMB half-life estimates have been reported from as low as 79–174 and 164–175 days^{40–42} to higher 349, 404, 427, >540 days.^{13,24,39,43,44} One study shows that the biodegradation half-life of EMB in sediments, collected directly below an active salmon farm, could be much longer than 200 days, potentially in the order of years, based on the reported depletion curves.¹³

Interestingly, the current study found that time since the last treatment and distance from the farm had the highest effect on EMB concentrations, while concentrations increased with latitude, meaning that EMB was generally lower at lower latitudes at similar distances, depths, and months since last treatment. This effect most probably could be explained by

different bottom water temperatures between the latitudes; at higher latitudes the mean bottom temperature was 4.9 °C and lower latitudes had a mean temperature of 7.8 °C. Hamoutene et al.⁴⁵ showed that EMB persistence in marine sediments is affected by temperature, organic matter, and the presence of antibiotics. They found that higher temperatures (10 °C vs 4 °C) increased EMB degradation, consistent with this study. However, their work and others highlight that multiple factors, not just temperature, influence EMB persistence. The variations of half-lives and decay can be explained by different conditions, (with aerobic (lower estimates) or anaerobic sediment conditions (higher estimates)), photoperiod, flow rates, organic matter, and different microbial communities and the impact of antibiotics. Furthermore, the authors do not have information about potential “off-label” use (meaning that higher quantities than is recommended by in-feed medicine producers could have been used), which could partly explain the differences related to occurrence of EMB in the environment.

Detection of low levels of teflubenzuron in sediments at sites, without recently reported treatments such as in this study, suggests inconsistencies in treatment reporting, dispersal from nearby farms, or long-term sediment persistence. BarentsWatch provides overviews of conducted treatments, while VetReg documents the prescriptions issued; however, not all treatments appear consistently registered, and prescriptions do not confirm administration. Cross-checking these databases, we found inconsistency that indicated discrepancies between prescribed and reported treatments.

Comparable findings have been reported elsewhere. A Norwegian screening of sediments at aquaculture sites in 2024 by Gundersen et al.⁴⁶ found teflubenzuron 10 years after treatment at one site. This study also detected elevated levels of delousing agents in sediments at the maximum distances of collection 1 km away and in benthic animals 400 m from the source site. The same study also found discrepancies in the reporting of delousing chemicals in Barents Watch and what was prescribed in VetReg. Parsons et al.⁴⁷ detected teflubenzuron in sediments near two farms where it had not been registered for use for four years, attributing this to either dispersal from nearby farms or sediment persistence. Similarly, Bloodworth et al.¹⁷ detected teflubenzuron around three salmon farms in Scotland and the Shetland Isles approximately four years after the last recorded use. At one farm in this study, high concentrations were observed despite no recent use at the farm or nearby use (>6 years), suggesting residues originated from older treatments or unregistered recent applications on the farm or nearby farms.

These findings highlight the long-term environmental persistence of teflubenzuron and emphasize the need for more consistent and complete treatment reporting to improve the accuracy of environmental monitoring and assessment.

Field studies show that EMB, teflubenzuron and diflubenzuron can persist and accumulate in sediments near salmon farms, depending on use, timing, and local conditions. However, data on EMB levels beyond 500 m are limited, even though models suggest harmful levels may spread over 1 km. Better understanding the wider distribution is vital to assess EMB impact on benthic ecosystems, especially in areas with dense aquaculture.

4.2. Impact of Environmental Parameters on Benthic Community Composition: Species Diversity and Sensitivity

The statistical results showed that benthic community composition (species richness) was most strongly affected by EMB, and secondarily by particle sediment size ($>63\ \mu\text{m}$) and TOC. EMB is known to affect species diversity negatively,¹⁷ but for some species or phyla, a threshold level in EMB concentration may exist before diversity is affected. The statistical modeling showed that the decrease in log number of all species with increasing log EMB is not linear but starts at a threshold level of EMB (roughly $\exp(-1.5) = 0.2\ \text{ng/g}$). This value corresponds well to the current environmental SEPA environmental quality standard of EMB at $0.272\ \text{ng/g}$, a value based on toxicological laboratory studies and sophisticated computer modeling. The value is set to be the maximum concentration of in-feed sea lice medicine in Scotland, at the mixing zone limit and beyond ($>100\ \text{m}$ from pens) to ensure good environmental status. This regulatory framework requires farm operators to manage their sites so that there is no significant adverse impact on the biodiversity of seabed life beyond the edge of the permitted mixing zone. SEPA will refuse to grant an authorization for proposed farm developments that are unlikely to achieve the current seabed quality standards.²¹

There are few field studies on effects of EMB on benthic fauna, however, Park⁴⁸ linked EMB field sediment concentrations to a reduction in the abundance and size of spot prawn (*Pandalus platyceros*) shortly after treatment. Telfer et al.⁴⁹ observed a decline in species diversity at a Scottish aquaculture site shortly after EMB treatment. However, this diversity returned to pretreatment levels within 12 months, and the study concluded that the observed impacts were not significant for a single treatment at one farm, attributing the effects to organic enrichment rather than EMB itself. Furthermore, the current study is consistent with the findings of Bloodworth et al.,¹⁷ who showed that EMB residues, following its use as an in-feed treatment for sea lice, are widely distributed in the benthic environment in the Shetland Isles. The Bloodworth et al. study revealed that these residues had a statistically significant impact on benthic ecology, particularly affecting the diversity and structure of benthic communities, with a pronounced effect on crustacean species. EMB was found to influence crustaceans even at very low concentrations, $0.763\ \mu\text{g/kg}$ wet weight sediment, highlighting its ecological implications in aquaculture environments.

Statistical analyses showed an unexpected positive relationship between TFB and species richness for Echinodermata caused by high echinoderm diversity observed at one specific site with the highest TFB concentrations. The same was found for overall species sensitivity due to observed high sensitivity values at the same site. This site consisted of steep rocky slopes and mixed substrate with many stones and rocks which are not adequately captured by the sampling method (grab). A high diversity of echinoderms—particularly *Asteroidea* and *Echinoidea*—was observed as well as many species typically considered sensitive to disturbance, most likely because the area provides favorable habitat conditions for these groups. Several of these recordings may represent a spillover effect from the surrounding rocky areas and are therefore not necessarily representative of the local conditions in the sediment itself. This suggests that, in such environments, grab sampling may not fully represent the actual community composition, and that

additional information or supplementary parameters should be considered when assessing species diversity and sensitivity in such conditions in the future (e.g., visual methods). In addition, many of the echinoderm, mollusk, and polychaete species recorded are not particularly sensitive to teflubenzuron, which acts by inhibiting chitin synthesis⁴⁷ and primarily affects crustaceans that possess a chitinous exoskeleton. In contrast, for example echinoderms lack chitin and have a calcareous endoskeleton, making them less susceptible to teflubenzuron exposure.

The current study, along with previous research, show that EMB is the primary factor for the decline in species richness and helps identify the threshold concentration at which this decline occurs (roughly $0.2\ \text{ng/g}$ EMB in sediment) corresponding well with the developed SEPA EQS ($0.272\ \text{ng/g}$ EMB in sediment) for preventing adverse impacts on the biodiversity from aquaculture farm operators.

4.3. Impact of Environmental Parameters on Crustacean Taxa

In the current study, EMB had the most pronounced negative effect on crustacean abundance and crustacean species richness. The decline in log abundance across all species with increasing log EMB is more pronounced for crustaceans compared to the general benthic community. This suggests that crustaceans are more sensitive to EMB, although a threshold level of EMB for species decline, such as for the overall benthic community, could not be precisely identified in the current study. A limitation in the current study is the detection level at $0.05\ \text{ng/g}$ for EMB in the sediment analyses. In practice, measurements were likely distributed between 0 and $0.05\ \text{ng/g}$. This means that the decrease in species richness (SR) might have been less steep if the highest SR values had been lower in EMB than $0.05\ \text{ng/g}$.

Nonetheless, it is clear that crustaceans are the most sensitive phylum in the current study. This is expected, as EMB is primarily designed to target parasitic crustacean copepods, such as the sea lice from the *Caligidae* family. EMB disrupts the gamma-aminobutyric acid (GABA) and chloride channels in crustaceans, causing impaired cell function, paralysis and eventually death.¹⁶ The findings of the current study corroborate the key findings reported by Bloodworth et al.¹⁷ and Wilding and Black,⁵⁰ who linked widespread crustacean impacts to the use of EMB using national operator return data and field survey data.

The majority of crustaceans identified in this study were amphipods, which are known to respond to disturbances in ways that vary depending on their lifestyle¹⁷ and references therein. In the current study, stations with sediment concentrations exceeding the SEPA threshold value of $0.272\ \text{ng/g}$ often showed a complete absence of crustaceans. At the few stations where crustaceans were present, they were found in very low numbers and consisted mainly of mobile epifaunal species, such as amphipods, ghost shrimp (*Caprellidae*), *Hippomedon* sp., *Westwoodilla cecula*, *Leptostraca* (e.g., sea flea *Nebalia* sp.), and cumacean (*Hemilamprops roseus*). Epifauna do not respond to discharges in the same way as infauna and they are able to move into areas of impact to feed opportunistically before withdrawing¹⁷ and references therein. Infaunal species on the other hand, typically exhibit low mobility, living within the sediment and feeding on detritus, with a focus on deposit-feeding or interstitial burrowing. These taxa were not found in sediments with EMB concentrations

above thresholds value of SEPA. These taxa include for example the burrowing amphipods (e.g., *Ampelisca* sp., *Byblis* sp., *Diastylis cornuta* and *Oedicerotidae*) and the burrowing isopods (e.g., *Asellota*). The findings in the current study are in accordance with the study of Bloodworth et al.¹⁷ which reported that the benthic species that were most sensitive to EMB concentrations shared several key characteristics: they were interstitial burrowers, obligate deposit feeders, and had low mobility. The current study further found a higher number of crustacean species in coarser sediments. This aligns with other studies showing that coarser sediments like sand offer better oxygenation, food availability, habitat complexity, and predator protection, making them more suitable for crustaceans. In contrast, fine sediments such as mud often become anoxic at depth, creating poor conditions for many species.^{51,52}

Benthic organisms play a vital role in the ecosystem by regulating carbon, nitrogen, and sulfur cycles, influencing water column processes, determining the distribution and fate of pollutants, contributing to secondary production, and promoting sediment stability and transport.⁵³ These organisms are essential for nutrient cycling, detritus decomposition, and serve as a key food source for larger organisms. For instance, bioturbation by these organisms can release EMB from sediments.⁵⁴ However, Bloodworth et al.¹⁷ reported that the slow and shallow bioturbation by opportunistic organisms, like the polychaete *Capitella capitata*, which are usually found at the edges of cages in enriched environments, may slow down the release of EMB from sediments. They argued that as a result, EMB could function as a long-term barrier to ecological succession, particularly for crustaceans that play important roles in bioturbation and ecosystem engineering.⁵⁵ Furthermore, they reported that it is still unclear whether this impact is caused by the toxic effects of consuming sediment with high EMB levels, or the long-term effects of eating sediment that lacks prey (interstitial crustacean meiofauna), a key part of the diet for vulnerable species. More research is needed to understand these processes.

Additionally, previous studies, such as those by Bloodworth et al.,¹⁷ have highlighted potential impacts of EMB on larger, more transient crustacean species. Waddy et al.,⁵⁶ for example, demonstrated that EMB interferes with the molting cycle of American lobsters (*Homarus americanus*). Given the economic importance of these species, continued surveillance and targeted research are essential to determine whether the observed findings extend to other crustacean taxa that are ecologically and commercially significant.

4.4. Toward More Sustainable Aquaculture Practices: Strengthening Oversight of EMB to Protect Benthic Biodiversity

The use of EMB in aquaculture for delousing purposes has been shown to significantly impact benthic biodiversity, particularly within 500 m of salmon farms, where harmful concentrations have been recorded. Field studies indicate that EMB is widespread, can persist and possibly accumulate in benthic sediments even long after its use. This may lead to a decline in species diversity and disruption of community structure at concentrations as low as 0.2 ng/g (in this study), a threshold close to the SEPA environmental quality standard of 0.272 ng/g (derived from laboratory studies). Crustaceans exhibit elevated sensitivity to EMB. Despite these findings, there is limited data on EMB concentrations beyond 500 m,

where dispersion models predict that harmful levels could extend up to 1 km from the farms.

Coastal biodiversity is already under threat from climate change and anthropogenic activities,⁵⁷ resulting in the disappearance of many species,⁵⁸ which in turn impacts vital ecosystem processes. For example, low sediment biodiversity can decrease the coastal ecosystem's stability and resistance, making it vulnerable to invasive species or other forms of disturbance.⁵⁹ In response to the growing threats to biodiversity and ecosystem health, international bodies such as the United Nations, the European Union, and Norway have implemented ambitious long-term strategies aimed at reversing ecosystem degradation and safeguarding biodiversity. Notable initiatives include the Norwegian Biodiversity Act,⁶⁰ the EU's Biodiversity Strategy for 2030,⁶¹ and the UN's 2030 Agenda for Sustainable Development.⁶² These frameworks emphasize the critical importance of biodiversity for maintaining healthy ecosystems and ensuring sustainable food systems.

Norway's current regulatory framework has significant gaps in managing chemicals like EMB, a commonly used delousing agent in aquaculture. Currently there are no standardized regulatory frameworks of consistent monitoring protocols in place for this substance. Although new regulations are now being developed,⁶³ which may require fish farms to conduct environmental assessments and sediment analyses for some delousing agents, it remains unclear how these results will be used or what actions will be taken if limits are exceeded. While some environmental quality standards (NEQS) exist for chemicals like flubenzurons in Norway, there are no clear safety thresholds for EMB. This lack of clarity, combined with the absence of cumulative impact assessments for multiple farms using EMB, highlights the need for improvement and more transparent regulation to protect biodiversity.

To mitigate these risks and support the long-term sustainability of aquaculture, it is essential to establish clear safety standards for the use of EMB in Norway and other salmon producing countries. Monitoring should extend beyond the immediate vicinity of aquaculture pens, at least 1 km from farm sites and account for the cumulative impacts of multiple farms in densely farmed areas. Furthermore, there is a need for improved consistency and completeness in treatment reporting to support accurate environmental monitoring and assessment. Without well-defined guidelines and robust monitoring, ecosystems remain vulnerable to ongoing degradation, threatening biodiversity and the future of more sustainable aquaculture. Immediate action is needed to close these gaps and promote more environmentally responsible practices across the industry.

■ ASSOCIATED CONTENT

Data Availability Statement

Dataset has been published on Zenodo.org, with DOI: <https://doi.org/10.5281/zenodo.17423989>.

Supporting Information

The Supporting Information is available free of charge at <https://pubs.acs.org/doi/10.1021/acs.est.5c11181>.

The methodology, data analyses, and supplementary figures; Appendix A: supplementary data; Appendix B: additional details on methodology and data analysis (PDF)

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M.A., K.S., A.M., A.L., and G.H.R. designed the study and field sampling. M.D. performed the oceanographic modeling. M.A., K.S., and G.H.R. analyzed the data and wrote the manuscript with the help of all the coauthors. V.R. performed the statistical testing. A.M., A.E., A.L., and D.B.S. reviewed and edited.

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REFERENCES

- (1) PwC Seafood Barometer. *Conditions For Sustainable Growth - An Unpredictable Future*. PricewaterhouseCoopers (PwC), 2023; <https://www.pwc.no/no/publikasjoner/2023-rapport-sjoematbarometeret.pdf>. Date of access: 25–08–2024.
- (2) Thorstad, E. B.; Forseth, T. Status for norske laksebestander i 2017. *Report from the Scientific Advisory Committee for Salmon Management No 10*. Scientific Council for Salmon Management 2017, 152.
- (3) Karlsen, Ø.; Asplin, L.; Finstad, B.; Sandvik, A. D.; Serrallinares, R. M.; Johnsen, I.; Askeland, N. R.; Berg, M.; Uglem, I.; Bjørn, P. A. (2018). Effekten av nasjonale laksefjorder på risikoen for lakselusinfestasjon hos vill laksefisk langs norskekysten-Sluttrapportering av ordningen med nasjonale laksefjorder. *Fisken Og Havet, HAVFORSKNINGSINSTITUTTET*, 2018; https://www.hi.no/resources/publikasjoner/fisken-og-havet/2018/effekten_av_nasjonale_laksefjorder_sluttrapport_nlf_korrigert_juni18.pdf.
- (4) Torrisen, O.; Jones, S.; Asche, F.; Guttormsen, A.; Skilbrei, O. T.; Nilsen, F.; Horsberg, T. E.; Jackson, D. Salmonlice-impact on wild

- salmonids and salmon aquaculture. *J. Fish Dis.* **2013**, *36* (3), 171–194.
- (5) Østevik, L.; Stormoen, M.; Evensen, Ø.; Xu, C.; Lie, K.; Nødtvedt, A.; Rodger, H.; Skagøy, A.; Manji, F.; Alarcon, M. Effects of thermal and mechanical delousing on gill health of farmed Atlantic salmon (*salmo salar* L.). *Aquaculture* **2022**, *552*, 738019.
- (6) Arnberg, M.; Refseth, G. H.; Allan, I. J.; Benedetti, M.; Regoli, F.; Tassara, L.; Sagerup, K.; Drivdal, M.; Nøst, O. A.; Evensen, A.; et al. Acute and Sublethal Effects of Deltamethrin Discharges from the Aquaculture Industry on Northern Shrimp (*Pandalus borealis* Krøyer, 1838): Dispersal Modeling and Field Investigations. *Environ. Sci. Technol.* **2023**, *57* (9), 3602–3611.
- (7) Helgesen, K. O.; Horsberg, T. E.; Stige, L. C.; Tarpai, A.; Norheim, K. The surveillance programme for resistance in salmon lice (*Lepeophtheirus salmonis*) in Norway 2023. *Surveillance program report*; Norwegian Veterinary Institute, 2024.
- (8) Norwegian Institute of Public Health, *Use of veterinary medicines in fish farming in 2012–2024*; 2022. <https://www.fhi.no/he/legemiddelbruk/fisk/bruk-av-legemidler-i-fiskeoppdrett/#midler-mot-lakselus>. Date of access: 25–08–2024.
- (9) Grefsrud, E. S.; Karlsen, Ø.; Andersen, L. B.; Grøsvik, B.; Hansen, P. K.; Husa, V.; Madhun, A. S.; Samuelsen, O.; Sandlund, N.; Solberg, M. F., et al. *Risk Report Norwegian Aquaculture 2024 — Risk Assessment*; Institute of Marine Research, 2024.
- (10) Campbell, W. C.; Fisher, M. H.; Stapley, E. O.; Albers-Schönberg, G.; Jacob, T. A. Ivermectin: a potent new antiparasitic agent. *Science* **1983**, *221*, 823–828.
- (11) Sevatdal, S.; Magnuson, Å.; Ingebritsen, K.; Haldorsen, R.; Horsberg, T. E. Distribution of emamectin benzoate in Atlantic salmon (*Salmo salar* L.). *J. Vet. Pharmacol. Ther.* **2005**, *28*, 101–107.
- (12) Burrige, L.; Weis, J. S.; Cabello, F.; Pizarro, J.; Bostick, K. Chemical use in salmon aquaculture: A review of current practices and possible environmental effects. *Aquaculture* **2010**, *306* (1), 7–23.
- (13) Benskin, J. P.; Ikononou, M. G.; Surridge, B. D.; Dubetz, C.; Klaassen, E. Biodegradation potential of aquaculture chemotherapeutants in marine sediments. *Aqua. Res.* **2016**, *47*, 482–497.
- (14) Strachan, F.; Kennedy, C. J. The environmental fate and effects of anti-sea lice chemotherapeutants used in salmon aquaculture. *Aquaculture* **2021**, *544*, 737079.
- (15) Grefsrud, E. S.; Karlsen, Ø.; Andersen, L. B.; Grøsvik, B.; Hansen, P. K.; Husa, V.; Madhun, A. S.; Samuelsen, O.; Sandlund, N.; Solberg, M. F., et al. *Risk Report Norwegian Aquaculture 2021 — Risk Assessment, Report*; Institute of Marine Research (IMR). 2021.
- (16) Burrige, L. E.; Hamilton, N.; Waddy, S. L.; Haya, K.; Mercer, S. M.; Greenhalgh, R.; Tauber, R.; Radecki, S. V.; Crouch, L. S.; Wislocki, P. G.; Endris, R. G. Acute toxicity of emamectin benzoate (SLICEtm) in fish feed to American lobster, *Homarus americanus*. *Aqua. Res.* **2004**, *35*, 713–722.
- (17) Bloodworth, J. W.; Baptie, M. C.; Preedy, K. F.; Best, J. Negative effects of the sea lice therapeutant emamectin benzoate at low concentrations on benthic communities around Scottish fish farms. *Sci. Total Environ.* **2019**, *669*, 91–102.
- (18) Fisheries and oceans Canada (DFO). Potential exposure and associated biological effects from aquaculture pest and pathogen treatments: anti-sea lice pesticides (part II). *Canadian Science Advisory Secretariat*. 2013; <https://waves-vagues.dfo-mpo.gc.ca/Library/350459.pdf>.
- (19) Page, F. H.; Burrige, L. Estimates of the effects of sea lice chemical therapeutants on non-target organisms associated with releases of therapeutants from tarped net-pens and well-boat bath treatments: a discussion paper *Canadian Science Advisory Secretariat Fisheries and oceans Canada (DFO) 2014* <https://waves-vagues.dfo-mpo.gc.ca/Library/359884.pdf>.
- (20) SEPA (Scottish Environment Protection Agency) *Interim Position Statement for Protecting the Water Environment until such Time as a Direction is Issued on an EQS in Relation to Emamectin Benzoate in Finfish Farm Regulation*; SEPA, 2021.
- (21) SEPA (Scottish Environment Protection Agency) . *New environmental quality standard (EQS) for emamectin benzoate*

(EmBz): consultation on the implementation timescales. SEPA, 2023. https://www.sepa.org.uk/media/594684/position_statement_embz-march-2023-approved.pdf. Date of access: 10–06–2024.

(22) Park, A. *biological effects of emamectin benzoate (SLICE®) on spot prawn (Pandalus platyceros)*. MSc thesis; University of Victoria, British Columbia, Canada, 2013.

(23) Kingsbury, M. V.; Hamoutene, D.; Kraska, P.; Lacoursiere-Roussel, A.; Page, F.; Coyle, T.; Sutherland, T.; Gibb, O.; Mckindsey, C. W.; Hartog, F.; Neil, S.; Chernoff, K.; Wong, D.; Law, B. A.; Brager, L.; Baillie, S. M.; Black, M.; Bungay, T.; Gaspard, D.; Hua, K.; Parsons, G. J. Relationship between in feed drugs, antibiotics, and organic enrichment in marine sediments at Canadian Atlantic salmon aquaculture sites. *Mar. Pollut. Bull.* **2023**, *188*, 114654.

(24) Ikononou, M. G.; Surridge, B. D. Ultra-trace determination of aquaculture chemotherapeutants and degradation products in environmental matrices by LC-MS/MS. *Int. J. Environ. Anal. Chem.* **2013**, *93* (2), 183–198.

(25) Langford, K. H.; Øxnevad, S.; Schøyen, M.; Thomas, K. V. Do Antiparasitic Medicines Used in Aquaculture Pose a Risk to the Norwegian Aquatic Environment? *Environ. Sci. Technol.* **2014**, *48*, 7774–7780.

(26) Sagerup, K.; Refseth, G. H.; Drivdal, M.; Allan, I. A.; Raantic, R.; Borge, E.; Blévinb, P.; Martins, S. E.; Carlsson, P.; Macken, A. L., et al. Floating versus sinking: environmental assessment of a novel medicinal feeding method for sea lice control in Aquaculture. 2025.

(27) Tucça, F.; Moya, H.; Pozo, K.; Borghini, F.; Focardi, S.; Barra, R. Occurrence of antiparasitic pesticides in sediments near salmon farms in the northern Chilean Patagonia. *Mar. Pollut. Bull.* **2017**, *115* (1), 465–468.

(28) Standard Norway. *Environmental Monitoring Of Fish Farming Facilities NS 9410. This Standard Describes a Method For Measuring And Monitoring Seabed Conditions At Aquaculture Facilities*. 2016; <https://standard.no/fagomrader/fiskeri-og-akvakultur/miljoovervaking-av-matfiskanlegg/>. Date of access: 2024–11–10.

(29) Aquaculture Stewardship Council ASC Salmon Standard (Version 1.3); Aquaculture Stewardship Council, 2019; https://www.asc-aqua.org/wp-content/uploads/2019/07/ASC-Salmon-Standard_v1.3_final.pdf. Date of access: 30–03–2025.

(30) International Organization for Standardization (ISO). *Water Quality: guidelines For Quantitative Sampling And Sample Processing Of Marine Soft-Bottom Macrofauna*. ISO, 2014; <https://www.iso.org/standard/54846.html>. Date of access: 10–11–2024.

(31) Norwegian Food Safety Authority *vetreg: veterinært Legemiddelregister*. 2024. Accessed 19. 02. 2024. <https://www.mattilsynet.no>.

(32) Fish health Database for the aquaculture sites, weekly overview including sea lice and treatments. BarentsWatch, <https://www.barentswatch.no/fiskehelse/2025/42>. Accessed: 06. 05. 2024.

(33) Horton, T.; Kroh, A.; Ah Yong, S.; Bailly, N.; Boyko, C. B.; Brando, S. N.; Gofas, S.; Hooper, J. N. A.; Hernandez, F.; Holovachov, O. et al. *World Register of Marine Species*; WoRMS Editorial Board, 2018.

(34) Norwegian Environment Agency. *Threshold Values For Classification Of Water, Sediment, And Biota (revised 30.10.2020)*. Norwegian Environment Agency: Oslo, 2023; <https://www.miljodirektoratet.no/globalassets/publikasjoner/m608/m608.pdf>.

(35) Chen, C.; Liu, H.; Beardsley, R. C. An unstructured grid, finite-volume, three-dimensional, primitive equations ocean model: application to coastal ocean and estuaries. *J. Atmos. Ocean. Technol.* **2003**, *20* (1), 159–186.

(36) Bruggeman, J.; Bolding, K. A general framework for aquatic biogeochemical models. *Environ. Model. Softw.* **2014**, *61*, 249–265.

(37) Burnham, K. P.; Anderson, D. R.; Huyvaert, K. P. AIC model selection and multimodel inference in behavioral ecology: some background, observations, and comparisons. *Behav. Ecol. Sociobiol.* **2011**, *65*, 23–35.

(38) Hamoutene, D.; Salvo, F.; Egli, S. N.; Modir-Rousta, A.; Knight, R.; Perry, G.; Bottaro, C. S.; Dufour, S. C. Measurement of aquaculture chemotherapeutants in flocculent matter collected at a

hard-bottom dominated finfish site on the south coast of Newfoundland (Canada) after 2 years of fallow. *Front Mar Sci.* **2018**, *5*, 1–11.

(39) Fisheries and Oceans Canada (DFO) Assessment of the Fate of Emamectin Benzoate, the Active Ingredient in SLICE®, near Aquaculture Facilities in British Columbia and its Effect on Spot Prawns (*Pandalus platyceros*). *Canadian Science Advisory Secretariat Science Advisory Report*. Fisheries and Oceans Canada, 2011.

(40) SEPA (Scottish Environment Protection Agency). *Fish Farm Advisory Group. Emamectin Benzoate Use In Marine Fish Farms: an Environmental Risk Assessment*. SEPA, 1999. <https://www.sepa.org.uk/media/299675/wrc-uc12191-03-review-of-environmental-quality-standard-for-emamectin-benzoate.pdf>. Date of access: 10–06–2024.

(41) SEPA (Scottish Environment Protection Agency). *Regulation and monitoring of marine cage fish farming in Scotland – A procedures manual, Attachment XI, Guidance on the use of Emamectin Benzoate at marine cage fish farms*. SEPA Version, 2004.

(42) Bright, D. A.; Bio, R. P.; Dionne, S. *Use of Emamectin Benzoate in the Canadian finfish aquaculture industry: review of environmental fate and effects*. 2005, UMA Engineering Ltd for Fisheries and Oceans Canada: Vancouver, BC, 2005.

(43) Hamoutene, D.; Salvo, F. Biodegradation of some aquaculture chemotherapeutants weathered in flocculent samples collected at hard-bottom sites in Newfoundland and Labrador (Canada). *Mar. Chem* **2020**, *224*, 103811.

(44) Ikononou, M. G. Assessment of the fate of Emamectin Benzoate, the active ingredient in Slice®, near aquaculture facilities in British Columbia and its effect on the Pacific spot prawn (*Pandalus platyceros*). *Canadian Science Advisory Secretariat Science Advisory Report*; Canadian Science Advisory, 2011; pp. 12.

(45) Hamoutene, D.; Kingsbury, M.; Davies, J.; Le, A.; Blais, D. R.; Gagnon, M. The persistence of Emamectin Benzoate in marine sediments with different organic matter regimes, temperature conditions, and antibiotic presence. *Mar. Pollut. Bull.* **2023**, *197*, 115714.

(46) Gundersen, C. B.; Reid, M.; Rundberget, T.; Bæk, K.; Dale, T.; Svendsen, F. S.; Åsnes, H. Ø.; Ndungu, K. *Screening Programme 2024. Niva Report*; miljodirektoratet, 2025. <https://www.miljodirektoratet.no/publikasjoner/2025/oktober-2025/screening-programme-2024/>.

(47) Parsons, A. E.; Samuelsen, O. B.; Johnsen, I. A.; Hannisdal, R.; Tjensvoll, T.; Husa, V. Distribution and Persistence of Diflubenzuron and Teflubenzuron in the Marine Environment Around Salmonid Aquaculture Facilities. *Front Mar Sci.* **2021**, *8*, 694577.

(48) Park, A. *biological effects of Emamectin Benzoate (SLICE®) on spot prawn (Pandalus platyceros)*. MSc thesis; University of Victoria: British, Columbia, Canada, 2013.

(49) Telfer, T. C.; Baird, D. J.; McHenry, J. G.; Stone, J.; Sutherland, I.; Wislocki, P. Environmental effects of the anti-sea lice (Copepoda: Caligidae) therapeutant Emamectin Benzoate under commercial use conditions in the marine environment. *Aquaculture* **2006**, *260*, 163–180.

(50) Wilding, T.; Black, K. SARF098: PAMP Refreshment Study – final report: A statistical analysis of sea - lice medicine use and benthic monitoring at Scottish marine salmon farms (2002 – 2014). *Scottish Aquaculture Research Forum*; Scottish Marine Institute, 2015.

(51) Hughes, R. N. Optimal foraging theory in benthic animals. *Oecologia* **1980**, *45* (1), 7–13.

(52) Lohrer, A. M.; Gibbs, M. T.; Hewitt, J. E. The effect of sediment grain size on benthic community structure. *Mar. Freshwater Res.* **2004**, *55* (1), 81–89.

(53) Snelgrove, P. V. R. Getting to the bottom of marine biodiversity: Sedimentary habitats: Ocean bottoms are the most widespread habitat on Earth and support high biodiversity and key ecosystem services. *BioScience* **1999**, *49* (2), 129–138.

(54) Stomperudhaugen, E. S.; Langford, K.; Schaanning, M.; Hylland, K. Release of emamectin from sediment: effects of oil, organic material or infauna? *J. Soils Sediments* **2014**, *14*, 1469–1478.

(55) Coates, D. A.; Alexander, D.; Herbert, R. J. H.; Crowley, S. J. Conceptual ecological modelling of shallow sublittoral sand habitats to inform indicator selection. *Marine ecological surveys Ltd - A report*

for the Joint Nature Conservation Committee JNCC Report No. 585. JNCC: Peterborough. 2016.

(56) Waddy, S. L.; Merritt, V. A.; Hamilton-Gibson, M. N.; Aiken, D. E.; Burridge, L. E. Relationship between dose of emamectin benzoate and molting response of ovigerous American lobsters (*Homarus americanus*). *Ecotoxicol. Environ. Safety* **2007**, *67* (1), 95–99.

(57) Holon, F.; Marre, G.; Parravicini, V.; Mouquet, N.; Bockel, T.; Descamp, P.; Tribot, A.; Boissery, P.; Deter, J. A predictive model based on multiple coastal anthropogenic pressures explains the degradation status of a marine ecosystem: Implications for management and conservation. *Biol. Conserv.* **2018**, *222*, 125–135.

(58) Pan, J.; Marcoval, M. A.; Bazzini, S. M.; Vallina, M. V.; De Marco, S. G. *Coastal Marine Biodiversity Challenges and Threats*; CRC Press: Boca Raton, 2013; pp. 43–67.

(59) Levin, L. A.; et al. The function of marine critical transition zones and the importance of sediment biodiversity. *Ecosystems* **2001**, *4* (5), 430–451.

(60) Lovdata Lov om forvaltning av naturens mangfold (Naturmangfoldloven) [Act on management of biodiversity]; Lovdata, 2001.

(61) European Commission EU Biodiversity Strategy For 2030: bringing Nature Back Into Our Lives European Commission 2020 https://environment.ec.europa.eu/strategy/biodiversity-strategy-2030_en

(62) United nations (UN). *Transforming Our World: the 2030 Agenda For Sustainable Development. Resolution Adopted By The General Assembly On 25 September*; United nations (UN), 2015. <https://sdgs.un.org/2030agenda>.

(63) Norwegian Ministry of Climate and Environment, *Law. Pollution Control Act (Act No. 6 of 13 March 1981). New pollution regulations, Pollution Regulations Chapter 34, and Pollution Control Act § 11, cf. § 16*; Norwegian Ministry of Climate and Environment, 1981.