



Physiological resilience of blue mussels (*Mytilus edulis*) to leachates of an anticorrosion paint commonly used on North Sea offshore wind farms

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ABSTRACT

As offshore wind energy undergoes rapid expansion, understanding the ecotoxicological risk of infrastructure-related chemical stressors, such as anticorrosion paint leachates, is vital for regional environmental management. This study investigated the acute physiological effects of leachates of paints commonly applied in North Sea offshore wind farms on the key bioindicator *Mytilus edulis*. Mussels were exposed for 14 days under three conditions: in direct contact with painted steel plates (direct exposure), near painted plates (indirect exposure), and in uncontaminated seawater (control). Physiological parameters including oxygen consumption, clearance rate, and ammonia excretion, were measured to derive the scope for growth (SFG), an integrative indicator of energy balance. Results showed no statistically significant difference in SFG or individual physiological rates across treatments. Mean SFG values (\pm SD) were 20.6 ± 12.2 , 21.2 ± 10.4 and 20.2 ± 11.5 J h⁻¹ g⁻¹ for control, direct and indirect exposures, respectively. These findings suggest that current anticorrosion coatings do not pose an immediate acute risk to adult *M. edulis*, even at concentrations exceeding expected offshore dilution. Although chronic exposure assessments and early life-stage studies incorporating molecular biomarkers remain critical gaps, this research provides essential baseline data to support environmental risk assessment and monitoring frameworks for the North Sea's offshore wind sector.

1. Introduction

Decarbonization of electricity generation is a global necessity, positioning wind energy as a cornerstone of the transition to carbon neutrality. With the global installed capacity of wind power reaching 1,174 Gigawatt, the contribution of offshore wind energy is rapidly accelerating and currently stands at 82.2 GW (approximately 7.1%) (Díaz and Guedes Soares, 2020; Global Wind Energy Council, 2025). The continuous expansion of offshore wind farms is primarily driven by their inherent operational advantages over onshore systems, specifically higher and more consistent wind speeds and minimal spatial conflicts.

Besides energy production, offshore wind farms provide socio-economic and ecological benefits, such as co-location with aquaculture and artificial reef effects that enhance biodiversity (Bela H. Buck et al., 2017; Di Tullio et al., 2018; Langhamer, 2012; Degraer et al., 2020) This multi-use approach is a central pillar of the European Union "Blue Growth" strategy, which aims to maximize the sustainable potential of marine and maritime sectors through the integration of renewable energy and marine food production (Spaniol and Rowland,

2022). However, corrosion of submerged steel support structures remains a major challenge, it is typically mitigated by cathodic protection and anticorrosion paints (Kirchgeorg et al., 2018). While environmental concerns have focused on noise (Popper et al., 2022; Tougaard et al., 2020), bird collisions (Brabant et al., 2015; Martin and Banks, 2023), and magnetic fields (Farr et al., 2021), chemical emissions from anticorrosion paints remain poorly understood. Mussels are among the earliest and most persistent colonizers of submerged artificial hard substrates, including ship hulls, port pilings, offshore mining platforms, and aquaculture infrastructure commonly coated with protective paints (Ergin and Ergin, 2021). This ecological behaviour renders mussels susceptible to chronic exposure to leachates from such paints, including trace metals and organic compounds (Chen et al., 2023). Although several North Sea bordering countries have regulations in place that prohibit biocidal agents in anticorrosion paints (Hengstmann et al., 2025) recent studies report complex mixtures of organic compounds leaching from anticorrosion paints (Alter et al., 2025), highlighting a critical knowledge gap.

The present study assesses the ecotoxicological effects of leachates

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from an anticorrosion paint commonly applied on North Sea offshore wind farms by evaluating the scope for growth (SFG) of exposed *Mytilus edulis* under controlled laboratory conditions. Mussels are widely used as bioindicators due to their sessile nature, high filter feeding and pollutant accumulation abilities (Beyer et al., 2017; Vereycken and Aldridge, 2023). Scope for growth, as a sensitive metric for sublethal stress, predicts growth potential by integrating energy acquisition and energy utilization (Vercauteren and Blust, 1999; Widdows and Johnson, 1988; Widdows and Staff, 2006). Additionally, as an instantaneous measure of energy status, SFG may act as a quantitative response to environmental stressors (Widdows and Staff, 2006). Previous research demonstrates its versatility in monitoring coastal ecosystems, including responses to summer mortalities (Tremblay et al., 1998) and contaminant levels (Olsson et al., 2004; Widdows et al., 2002). Laboratory studies have applied SFG to quantify the effects of stressors such as food limitation (Helson and Gardner, 2007), pharmaceuticals (Ericson et al., 2010), hypoxia (Sanders et al., 2014), heatwaves (Xu et al., 2023), and diet quality (Liénaert et al., 2023).

We hypothesized that chemical leachates from the paint would impair mussel bioenergetics, by reducing clearance rates and elevating metabolic and catabolic costs, thereby lowering the scope for growth. Such decline would reflect compromised energy balance and potential fitness consequences. To evaluate this, we quantified SFG in mussels subjected to direct and indirect paint exposure.

2. Material and methods

2.1. Test organism

Adult mussels ($n = 1200$; mean shell length 58.4 ± 3.5 mm) were collected from Zeeboerderij Westdiep, [Belgian coast: $51^{\circ}09.924'N$, $2^{\circ}37.747'E$] and transported to the University of Antwerp. Mussels were cleaned of epibionts and stocked in high-density polyethylene (HDPE) tanks ($n = 2$, at 1.5 mussels L^{-1}) in artificial seawater with salinity 33 PSU. Seawater was prepared by dissolving 35.75 $g L^{-1}$ hw-Marinemix Sea salt (Wiegandt GmbH, Germany) in deionized water (Eurowater Belgium NV). Following gradual temperature reduction from $16^{\circ}C$, mussels were acclimated at $13^{\circ}C$ for 14 days at under a 12 h/12 h light/dark regime. Feeding was ensured with freeze-dried microalgae *Tetraselmis chuii* (TetraPrime C, Proviron, Belgium) at $25,000$ cells mL^{-1} twice weekly. Nitrogenous waste levels were monitored daily and maintained below 0.25 $mg L^{-1}$ and 5 $mg L^{-1}$ for ammonia and nitrite respectively and water was renewed weekly.

2.2. Experimental set-up and coupon exposure

Steel plates (coupons): $10 \times 20 \times 0.3$ cm coated with offshore anticorrosion paint (Interthane 990, RAL1003 (primer 1: Interzone 954 grey, primer 2: Interzone 954 white, top: Interthane), International) commonly applied on offshore wind farms of the North Sea were submerged in seawater in glass aquaria (at 1 coupon per 20 L). Coupons were suspended on both sides by two glass petri dishes for uniform water circulation (See supplementary material S. 2.2 for details). Two exposure scenarios were tested: Direct exposure, with mussels placed directly on the coupon; Indirect exposure, (mussels adjacent to the coupon) and Control (mussels in seawater without a coupon). Three mussels per treatment were tested in triplicate over 1, 2, 3, 7 and 14 days of exposure. Mussels were fed every 48 h with *T. chuii* at $25,000$ cells mL^{-1} . Fecal pellets were removed daily and full water exchange was done every 3 days with preconditioned seawater (one coupon per 20 L).

The chemical profile of the leachate used in the present study was established based on a non-target screening (NTS) performed in a parallel study utilizing identical paint formulations and experimental conditions (Alter et al., 2025). Briefly, water samples exposed to the paint coupons for a 14-day duration were analysed via comprehensive two-dimensional gas chromatography-mass spectrometry (GCxGC-MS).

This previous characterization identified 233 putative chemical compounds, providing a reference “chemical fingerprint” of the substances leaching from the painted surfaces. Given the identical experimental set-up, additional identification or quantification of individual constituents was not conducted in the present study. Consequently, the leachate exposure scenario was conceptually based on the complex chemical suite characterised in the parallel study (Alter et al., 2025).

2.3. Measurement of physiological parameters

After each exposure period, clearance rate (CR), respiration, ammonia excretion, and scope for growth (SFG) were determined with slight modifications following (Widdows and Staff, 2006).

2.3.1. Clearance rate

Clearance rate (CR) was determined as the volume (L) of water cleared of *T. chuii* cells per hour. Algae food stock solution was prepared by mixing 1 g of freeze-dried *T. chuii* per litre filtered ($0.45 \mu m$) artificial seawater. Subsequently, mussels were independently placed in 1 L glass filtration chambers spiked with *T. chuii*. ($25,000$ cells mL^{-1}). Each chamber was equipped with a cylindrical magnetic stirrer bar (20×6 mm) and placed on a digital magnetic stirrer (Velp Scientifica, Italy) at 700 rpm for homogenous suspension of algae cells. At every 15 min for 90 min, 10 mL water samples were collected and analyzed for cell concentration using a Coulter Multisizer 3 (Beckman Coulter, Inc © 1990–2001). The CR was calculated from the decline in \log_e algae cell concentration over time, using Eq.1.

$$CR (L h^{-1}) = (Vol) \times (\log_e C_1 - \log_e C_2) / \text{time interval (h)} \quad (1)$$

Where: Vol is the volume of water (L), C_1 and C_2 are initial and final cell concentrations. Maximum CR (CR_{max}) was defined as the highest CR observed over a 15 min interval. The CR was standardized to 1 g dry weight converted into energy equivalent using Eq.2. (See supplementary materials. S 2.3.1 for details).

2.3.2. Respiration rate

After CR measurements (Section 2.3.1), mussels were transferred to filtered seawater identical to their respective exposure medium and left undisturbed for 60 min. Thereafter, they were transferred to individual 800 mL air-tight glass respirometers filled with air saturated, filtered seawater. A blank respirometer served as control for background respiration. Each respirometer was equipped with a polarographic oxygen electrode (WTW, Oxi 3310, Germany) recording dissolved oxygen and temperature every 15 min for 90 min.

$$\text{Rate } (\mu mol O_2 h^{-1}) = [C(t_0) - C(t_1)] \times (V_r) \times 60 / (t_1 - t_0) \quad (3)$$

Where t_0 , t_1 are start and end time (min), $C(t)$ is oxygen concentration ($\mu mol O_2 L^{-1}$) at time t ; and V_r is the volume of respirometer.

The measured oxygen uptake rates were standardized to 1 g dry using the allometric equation [Eq.7] and correspondingly converted into energy equivalents [Eq. 4] (See S 2.3.2).

2.3.3. Excretion rate

Ammonia excretion was determined from water samples (5 mL) collected before and after respiration measurements. Samples were analysed spectrophotometrically at 640 nm (Spectronic ® 20 GenesysTM, USA) using the salicylate-hypochlorite method (Bower et al., 1980; Le and Boyd, 2012). Ammonia excretion rate was calculated using Eq. 5.

$$\text{Rate } (\mu mol NH_3-N h^{-1}) = (([C(t_0) - C(t_1)] / MW) \times (V_r) \times 1000 \times 60 / (t_1 - t_0)) \quad (5)$$

Where t_0 , t_1 are start and end time (min), $C(t)$ is the ammonia concentration ($\mu mol NH_3-N L^{-1}$), V_r is the respirometer volume; and MW is Molar mass of ammonia ($gmol^{-1}$). Excretion rates were standardized to

1 g converted into energy equivalents Eq.6 (See. S 2.3.3).

2.3.4. Scope for growth

Scope for growth was evaluated following (Widdows and Staff, 2006). Prior to SFG calculation, physiological rates were standardized to 1 g dry weight using allometric equation Eq.7

$$Y = aX^b \quad (7)$$

Where Y = physiological rate; X = dry body mass (g), a is the intercept, and b is the weight exponent. Values of b were clearance rate (CR_{max}) = 0.61, oxygen consumption rate = 0.42, and ammonia excretion = -0.3468.

SFG was then computed as Eq.8

$$\text{SFG (J h}^{-1} \text{ g}^{-1}) = \text{Energy absorbed (J h}^{-1} \text{ g}^{-1}) - (\text{Energy respired} + \text{Energy excreted}) \text{ (J h}^{-1} \text{ g}^{-1}) \text{ [Eq. 8].}$$

2.3.5. Condition index

Condition index (C.I), an indicator of mussel health and overall physiological status was calculated as a ratio of soft tissue mass to shell size (Riisgård et al., 2014):

$$\text{C.I.} = \text{tissue dry weight (mg)} / \text{shell length (cm)}^3 \quad (9)$$

2.3.6. Statistical data analysis

To assess the influence of paint exposure and duration on mussel physiological performance, we utilized a linear modelling approach. Prior to analysis, the condition index (C.I) was standardized (Z-scaled) to facilitate model convergence and allow for the comparison of effect sizes. Initially Linear Mixed-Effects Models (LMMs) were fit for all physiological parameters (CR_{max} , respiration rate, ammonia excretion rate and SFG) using the *lme4* package (RStudio). Treatment (Control, Direct, Indirect), Day (1,2,3,7,14), and their interaction (Treatment x Day) were defined as fixed effects, with C.I included as a covariate. The tank was included as a random intercept to account for potential nesting within the experimental setup. For CR_{max} , ammonia excretion rate and SFG, the LMM yielded singular fits, implying that the variance attributed to the random effect (tank) was negligible, consequently, these were simplified to standard Linear Models (LMs), with identical fixed effects and covariate structure to avoid overfitting.

Respiration rate was analysed using a mixed model with random intercept (tank) retained and restricted maximum likelihood (REML) in the *lmerTest* package. The respiration rate data were normalized using Yeo-Johnson transformation ($\lambda = 0.56$; R package *bestNormalize*) prior to analysis to meet the model assumptions (Riani et al., 2023; Yeo and Johnson, 2000). Model selections for LMs and LMMs were based on Akaike Information Criterion (AIC) (Müller et al., 2013). Model assumptions were assessed using Shapiro-Wilk tests, for normality and Levene's test for homoscedasticity and post hoc comparisons used Turkey marginal means.

Differences in C.I among treatments were tested (Kruskal-Wallis χ^2 (2) = 0.23, $p = 0.893$). Significance of fixed effect was assessed via Type II Wald χ^2 tests; model fit was evaluated using marginal (R^2_m) and conditional (R^2_c) coefficients of determination (Nakagawa and Schielzeth, 2013). A sensitivity analysis test was conducted to assess the robustness of non-significant ($P > 0.05$) results across-the-board the tested physiological parameters and to distinguish whether these outcomes reflected a true absence of effect or limitations in statistical sensitivity. Estimated marginal means (EMMs), 95% confidence intervals (CIs), and standardized effect sizes (Cohen's d) were derived using *emmeans* package, with d calculated using respective model's residual standard deviations (σ) and equivalent degrees of freedom. Correspondingly, the minimum detectable effect (MDE): defined as the smallest change in each physiological parameter that the experimental design could reliably resolve from background variability. The MDE was

estimated using the *pwr* package, incorporating sample size ($n = 9$ per treatment per sampling day), $\alpha = 0.05$ and parameter-specific residual standard deviations ($\sigma = 3.33, 1.73, 0.53, \text{ and } 10.89$) for clearance rate, respiration rate, ammonia excretion rate and scope for growth, respectively and a theoretical power set at 80%. The MDE was initially expressed as standardized effect size (Cohen's d), and subsequently converted to original measurement scale by multiplying the standard value by the corresponding standard deviation (σ) i.e., $\text{MDE} = d \times \sigma$. Additionally, a retrospective power analysis was conducted for scope for growth, treated as an integrative index of all the measured physiological parameters. The power analysis for scope for growth was benchmarked against a biologically relevant threshold $15 \text{ J g}^{-1} \text{ h}^{-1}$, representing the transition from a 'healthy' to a 'moderately stressed' state in *Mytilus spp* (Bellas et al., 2014). The power analysis revealed that with a sample size of $n = 9$ mussels per treatment per day and observed residual variance ($\sigma = 10.89$), the experimental design achieved 78.3% (approx. 80%) power to detect a shift of $15 \text{ J g}^{-1} \text{ h}^{-1}$ in SFG, with an MDE of $15.3 \text{ J g}^{-1} \text{ h}^{-1}$ at the 80% power convention. All statistical analyses were performed using RStudio (R Core Team, version 4.5.1), with a significance level of $\alpha = 0.05$. Results are presented as mean \pm SD (See S.2.3.6 for details).

3. Results

3.1. Maximum clearance rate

The maximum clearance rate in *Mytilus edulis* varied over the 14-day exposure, with notable individual variability across treatment (Fig. 1 A). Median values and interquartile ranges (IQR) showed marked overlap among the control, direct exposure, and indirect exposure groups, indicating no consistent treatment effect. A fixed-effects linear model revealed no significant effect of direct ($\beta = 0.08, p = 0.911$) or indirect exposure ($\beta = 0.53, p = 0.45$) relative to controls. Clearance rate was significantly lower on day 7 than day 1 across all treatments ($\beta = 2.03, p = 0.032$), suggesting a transient decline. Condition index was a significant negative predictor of clearance rate ($\beta = -0.94, p = 0.003$), indicating reduced clearance rate in mussels with better physiological condition (Fig. 1B; Supplementary Fig S1). The model was statistically significant ($F_{7,127} = 2.70, p = 0.012, \text{ Adjusted } R^2 = 0.08$), though it explained a modest proportion of the variance.

3.2. Respiration rate

Respiration rate of *Mytilus edulis* remained stable across treatments over 14 days (Fig. 2 A). A linear mixed-effects model revealed no significant effect of direct ($\beta = -0.36, p = 0.33$) or indirect exposure ($\beta = -0.04, p = 0.92$) compared to controls. Median values and IQRs overlapped across all groups, indicating consistent metabolic responses. Considerable individual variability and outliers were observed, representing heterogeneity in metabolic activity. Condition index was a significant negative predictor ($\beta = -0.59, t_{126} = -3.62, p < 0.001$), with high condition mussels exhibiting lower respiration rates (Fig. 2B; Supplementary Fig S2).

3.3. Ammonia excretion rate

Ammonia excretion in *Mytilus edulis* was not significantly affected by paint exposure (all $p > 0.05$), with overlapping medians and IQRs across treatments (Fig. 3). A significant temporal effect was observed, with reduced excretion on day 3 ($\beta = -0.59, p < 0.001$) and a marginal decrease on day 2 ($p = 0.08$) relative to day 1. Mussel condition index was not a significant predictor of ammonia excretion ($p = 0.4$). Like other metabolic parameters, individual variability and outlier were evident (Fig. 3).

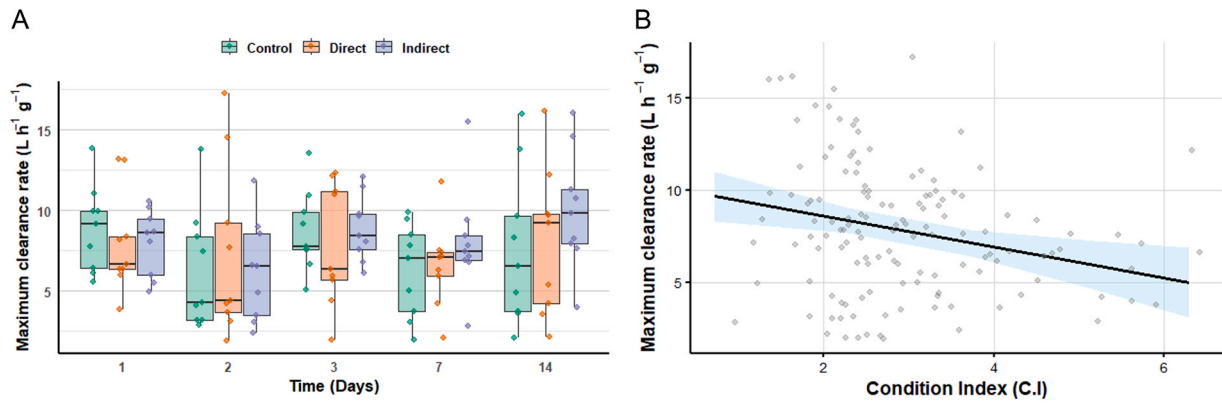


Fig. 1. (A). Maximum clearance rate ($L h^{-1} g^{-1}$) of mussels exposed to anticorrosion paint; control (green), direct (orange) and indirect (purple) over 14 days ($n = 9$ mussels per treatment per day). Boxes show the interquartile range (IQR), horizontal lines indicate the median, and whiskers extend to $1.5 \times$ IQR; Jittered points show individual replicates. (B) Relationship between predicted clearance rate and the condition index. The solid line represents the fitted model; shaded area indicates 95% confidence interval pooled across treatments.

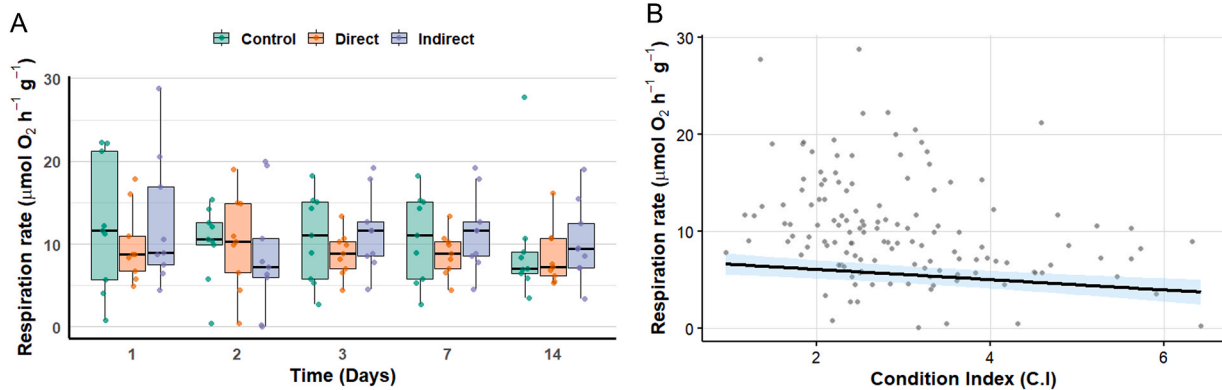


Fig. 2. (A). Respiration rate ($\mu mol O_2 h^{-1} g^{-1}$) of mussels exposed to anti-corrosion paint; control (green), direct (orange) and indirect (purple) over 14 days ($n = 9$ mussels per treatment per day). Boxes show the interquartile range (IQR), horizontal lines indicate the median, and whiskers extend to $1.5 \times$ IQR; Jittered points show individual replicates. (B) Relationship between predicted respiration rate and the condition index. Solid line shows the fitted model; shaded area indicates 95% confidence interval pooled across treatments.

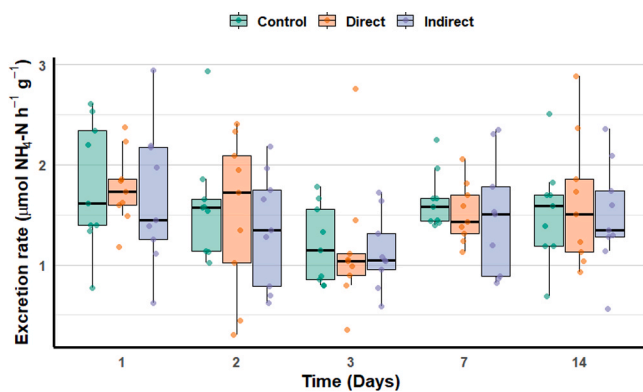


Fig. 3. Ammonia excretion rate ($\mu mol NH_4-N h^{-1} g^{-1}$) of mussels exposed to anti-corrosion paint; control (green), direct (orange) and indirect (purple) over 14 days ($n = 9$ mussels per treatment per day). Boxes denote the interquartile range (IQR), horizontal lines indicate the median, and whiskers extend to $1.5 \times$ IQR; Jittered points show individual replicates.

3.4. Scope for growth

Scope for growth (SFG) in *Mytilus edulis* was not significantly influenced by paint exposure. Fixed-effect linear modelling revealed that

neither direct exposure ($\beta = 1.30$, $p = 0.57$) nor indirect exposure ($\beta = 1.56$, $p = 0.49$) elicited measurable deviations in bioenergetic status compared to control individuals. Median SFG values and IQR exhibited substantial overlap across treatments and time points (Fig. 4 A), further demonstrated by overlapping effect sizes and 95% confidence intervals (Table 1). Most observations converged within a range of $10\text{--}35 J g^{-1} h^{-1}$, although notable individual plasticity was evident. This was highlighted by maximum individual values reaching approximately $52 J g^{-1} h^{-1}$ on Day 2, suggesting high stochastic variation within the population. While a marginal decline in SFG was observed on day 7 relative to day 1 ($\beta = -5.43$, $p = 0.078$), this trend did not reach statistical significance. The Condition index was a significant negative predictor of energy balance ($\beta = -2.39$, $p = 0.018$), where individuals with a lower condition index exhibited paradoxically higher SFG (Fig. 4B; Supplementary Fig S3). Overall, the model accounted for a modest proportion of the observed variance ($F_{7,127} = 2.22$, $p = 0.036$, adjusted $R^2 = 0.06$). These results suggest that variation in physiological state, rather than short-term paint exposure, was the dominant driver of SFG.

4. Discussion

Chemical emissions from offshore wind farms (OWFs) have recently gained increasing attention due to the rapid expansion and growing significance of offshore wind energy infrastructure (Ebeling et al., 2023; Hengstmann et al., 2025; Lazuga, 2024; Wang et al., 2023). Against this

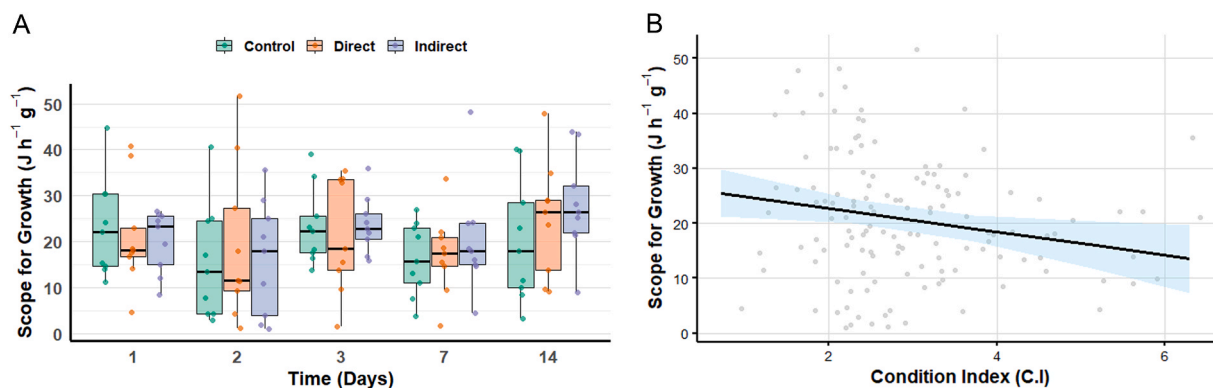


Fig. 4. (A). Scope for growth ($\text{J h}^{-1} \text{g}^{-1}$) of mussels exposed to anti-corrosion paint; control (green), direct (orange) and indirect (purple) over 14 days ($n = 9$ mussels per treatment per day). Boxes denote the interquartile range (IQR), horizontal lines indicate the median, and whiskers extend to $1.5 \times \text{IQR}$; Jittered points show individual replicates. (B) Relationship between predicted respiration rate and the condition index. Solid line represents the fitted model; shaded indicates 95% confidence interval pooled across treatments.

Table 1

Standardized effect sizes (Cohen's d), 95% confidence intervals (CIs) for pairwise comparisons and minimum detectable effect (MDE; calculated for $n = 9$, $\alpha = 0.05$, power = 0.80) for mussel physiological parameters. Clearance rate (CR; $\text{L g}^{-1} \text{h}^{-1}$), respiration rate (RR; $\mu\text{mol O}_2 \text{h}^{-1} \text{g}^{-1}$), ammonia excretion rate (AE; $\mu\text{mol NH}_4\text{-N h}^{-1} \text{g}^{-1}$) and scope for growth (SFG; $\text{J h}^{-1} \text{g}^{-1}$). MDE values are constant within each physiological parameter and are shown once per group.

Parameter and MDE	Comparison	Cohen's d (95% CI)
CR 4.68	Control vs Direct	-0.02 (-0.44–0.39)
	Control vs Indirect	-0.16 (-0.58–0.26)
	Direct vs Indirect	-0.14 (-0.56–0.28)
RR 2.10	Control vs Direct	0.21 (-0.33–0.74)
	Control vs Indirect	0.02 (-0.52–0.56)
	Direct vs Indirect	-0.19 (-0.73–0.35)
AE 0.75	Control vs Direct	0.07 (-0.34–0.49)
	Control vs Indirect	0.23 (-0.19–0.65)
SFG 15.31	Direct vs Indirect	0.16 (-0.26–0.58)
	Control vs Direct	-0.12 (-0.54–0.03)
	Control vs Indirect	-0.14 (-0.56–0.28)
	Direct vs Indirect	-0.02 (-0.44–0.40)

background, the present study assessed the biological effects of OWF-related coatings using painted coupons treated with Interthane 990, one of the most widely applied anticorrosion paint in North Sea OWFs.

Our results indicate that key, physiological parameters: clearance rate, respiration, and ammonia excretion of *Mytilus edulis* showed no significant changes following short-term exposure to anticorrosion paint. The observed responses were characterized by pronounced inter-individual variability and transient temporal fluctuations, suggesting that intrinsic physiological traits and temporal dynamics were the dominant drivers of variability and may have masked potential subtle effects under the tested conditions. The absence of significant differences in SFG aligns with previous reports that its sensitivity in mussels is greatest when pollutant concentrations exceed critical thresholds (Richardson et al., 2008; Widdows and Johnson, 1988). Leachate concentrations from paint coupons likely remained below levels required to elicit measurable physiological stress (Widdows et al., 2002). High natural inter-individual variability in mussels (Fuentes-Santos et al., 2018; Riisgård et al., 2011) may further obscure subtle responses, particularly under benign stress conditions (Tanner and Dowd, 2019). Minor, short-lived declines in clearance rate and SFG at day 7 followed by recovery suggest transient physiological

adjustments, possibly due to compensatory mechanisms restoring energy balance. However, the subtle nature of these changes underscores the limited sensitivity of whole organism endpoints for detecting early or low-level toxicological effects. Incorporating cellular biomarkers such as oxidative stress and gene expression could reveal responses preceding organismal changes (Bellas et al., 2014). Similar compensatory patterns have been observed in *Perna perna* exposed to DCOIT (4,5-dichloro-2-octyl-4-isothiazolin-3-one) where haemocyte counts recovered within 96 h (Fonseca et al., 2020).

Scope for growth is primarily governed by clearance rate (feeding), which accounts for approximately 70% of the total energy budget, whereas respiration and ammonia excretion together contribute the remaining 30% (Bellas et al., 2014). Because clearance and respiration processes are tightly linked through the multifunctional gills whose performance depends on the valve opening behaviour, any factor that disrupts valve activity can in turn, modify these core physiological processes (Bertolini et al., 2022; Riisgård et al., 2011). Sub-lethal xenobiotic or environmental stressors capable of altering valve dynamics therefore have the potential to influence overall mussel energy balance. For instance, increased filtration driven by enhanced valve opening may serve as an adaptive compensation to elevated metabolic demands during environmental perturbations demonstrating the physiological plasticity and stress-coping capacity characteristic of many bivalves (Ericson et al., 2010; Li et al., 2024). In the present study, however, consistent with the absence of statistically significant effects, a parallel experiment detected no notable changes in valve gape opening during the same exposure period (Alter et al., 2025). This suggests that any potential compensatory valve adjustments, if they occurred, were likely rapid, subtle and transient rather than sustained. Accordingly, the absence of a significant changes in valve gape behavior, suggests that mussels' core ecosystem functions may remain stable under the tested exposure conditions. Processes such as biofiltration and benthic-pelagic coupling which are vital for maintaining water clarity and nutrient cycling in the marine ecosystems (Beyer et al., 2017), are unlikely to be immediately impaired. However, these findings should be interpreted with caution as subtle or cumulative effects on physiological and behavioral responses may not be detectable under short-term laboratory conditions. Conversely mussels in situ, particularly those associated with offshore wind infrastructures, are exposed to dynamic environmental factors that may induce moderate to severe sub-lethal effects. Furthermore, the life span of offshore wind installations (often exceeding two decades) may facilitate the accumulation of chemical leachates to biologically relevant levels. Consequently, long-term, field-based monitoring studies are also required to unravel potential implications for population dynamics and ecosystem functioning.

Accordingly, we evaluated whether limited sample size may have

constrained effect detection, since inadequate sampling can result in underestimation of pollutant impacts and diminish the sensitivity to critical environmental or biological thresholds. (Babitsch and Sundermann, 2020). Nevertheless, despite the modest sample size ($N = 9$ per treatment group per time point), we carefully evaluated the sensitivity of our experimental design using a multi-tiered approach. For example, a retrospective sensitivity analysis indicated a minimum detectable effect (MDE) for scope for growth of $15.31 \text{ J g}^{-1} \text{ h}^{-1}$ at 78.3% power ($\alpha = 0.05$). This MDE value follows within range for established physiological (SFG) threshold for *Mytilus* spp, where a decline of approximately $15 \text{ J g}^{-1} \text{ h}^{-1}$ represents the transition from a healthy state to moderate stress (Bellas et al., 2014). Applying an identical sensitivity framework yielded MDE values of $4.68 \text{ L g}^{-1} \text{ h}^{-1}$, $0.75 \text{ } \mu\text{mol NH}_4\text{-N g}^{-1} \text{ h}^{-1}$, $2.1 \text{ } \mu\text{mol O}_2 \text{ g}^{-1} \text{ h}^{-1}$ for clearance rate, ammonia excretion and respiration rate respectively (Table 1). This suggests that our study was sufficiently powered to detect biologically meaningful shift across all tested physiological parameters when present. Furthermore, in line with best practice (Mair et al., 2020), we reported 95% confidence intervals (CIs) and standardized effect sizes (Table 1), all of which were negligible (Cohen's $d \leq 0.2$) with narrow CI centered near Zero. Notably, the minimum detectable effects (MDEs) exceeded the observed effect sizes, indicating that the absence of statistical significance is not a consequence of limited power or higher individual variability but rather these results suggest absence of measurable physiological effects under the tested exposure conditions. Therefore, in the current context of offshore wind energy development, these findings suggest that blue mussel populations, key inhabitants of submerged wind infrastructure may maintain physiological integrity despite the potential exposure to chemicals leachates from the corrosion protection paints. However, this conclusion is limited to the exposure levels and duration examined in the present study.

Intriguingly, the condition Index (C.I) was negatively associated with the mussel physiological performance (Fig. 4B), where individuals with lower C.I exhibited higher SFG. Although this pattern should be cautiously interpreted, the observed inverse relationship may reflect a complex trade-off in energy allocation potentially linked to reproductive phenology. For example, mussels in advanced stages of gametogenesis typically exhibit higher tissue mass and a correspondingly elevated C.I. However, the metabolic maintenance of ripe gonadal tissue imposes a significant respiratory demand, which can suppress the energy available for somatic growth. Conversely, the higher SFG corresponding with low condition individuals may suggest a compensatory physiological strategy, where depleted individuals maximize energy acquisition (i.e. via increased CR) and minimize metabolic demands to facilitate rapid tissue recovery or energy storage following spawning events (Beyer et al., 2017; Gilek et al., 1992). Therefore, the potential effects of paint exposure might have been masked by strong intrinsic mechanisms in addition to the higher individual variability. Our findings suggest that for future ecotoxicological assessments, the use of individuals within a narrower range of condition index is vital to improve the statistical power required to discern subtle anthropogenic stressors from baseline physiological noise.

Additionally, paints used in several anthropogenic activities represent a complex chemical source releasing mixtures of inorganic and organic compounds (Chen et al., 2023; Kirchgeorg et al., 2018; Soroldoni et al., 2017). Consequently, resident marine biota are likely exposed to complex mixtures capable of inducing a cascade of effects, from sublethal physiological perturbations in individuals to broad-scale disruptions of ecosystem structure and function. Our design complements a non-target analysis that identified 233 putative compounds, nine of which were unequivocally linked to leachates from the same anticorrosion paint used in the current study (Alter et al., 2025). This chemical fingerprint spanned five classes: Phenols (2,4-Dibromophenol, 2,6-Dibromophenol), Alcohols (1-(4-Methylphenyl) ethanol, 2-(4-Methylphenyl) propan-2-ol), Ketones (2,3-Dihydroinden-1-one), Halogenated Aromatic Hydrocarbon (1-Bromo-2,4-dimethylbenzene,

2-Bromo-1,3-dimethylbenzene, 2-Bromo-2,3-dimethylbenzene), and Aromatic Hydrocarbons (1,2-Dihydroacenaphthylene and 1-(4-Methylphenyl) benzene) (Alter et al., 2025). A maximum in abundance of these compounds can be expected after 6–8 days of exposure, but bio-accumulation of these compounds is unlikely to occur due to low xlogp values (<2.5) (Alter et al., 2025). Moreover, the screening was limited to GC-MS detectable compounds, excluding inorganic and organic compounds that are not detectable by GC-MS. This highlights the need for future research studies combining characterization and quantification of the organic and inorganic chemical compounds leaching from the anticorrosion paints (such as Interthane 990, RAL1003), to reassess their safety amidst global offshore wind expansion.

We acknowledge that several coating systems are currently used on North Sea offshore wind farms (e.g., Hempthane) and that anticorrosion paints represent only one of several potential sources of chemical emissions (Hengstmann et al., 2025). Additional continuous inputs may arise from other corrosion protection systems, while discontinuous emissions can result from maintenance activities, fire drills, sewage treatment, or accidental spills (Hengstmann et al., 2025; Kirchgeorg et al., 2018; Wang et al., 2023). Given the anticipated large-scale expansion of OWFs and the persisting gaps in knowledge and regulations regarding OWFs-related chemical emissions, and their ecological effects, there is a clear need to Advance research on both acute and chronic exposure scenario (Hengstmann et al., 2025). Improving this knowledge base is essential to support the development of robust environmental risk assessments and adaptive management strategies, and to inform policy implementation under regional frameworks such as the EU Marine Strategy Framework Directive, (Descriptor 8 on, contaminants) alongside OSPAR objectives (Vasilakopoulos, Boschetti, 2022). Such efforts are particularly vital in the North Sea, a region of high ecological and socioeconomic value already subject to multiple anthropogenic pressures, where sustainable offshore energy development needs to be balanced with long-term ecosystem protection and conservation.

5. Conclusion

In summary, our findings suggest that exposure to anticorrosion paint leachates characteristic of offshore wind infrastructure may not cause significant acute alterations in the energy balance of adult *Mytilus edulis*. The experimental design represents a conservative laboratory exposure scenario (i.e., 418 cm^2 of paint surface area per 20 L seawater), which may exceed the average dilution potential of the dynamic North Sea environment. However, in the absence of in situ flux or concentration data, these findings should be interpreted with caution, and extrapolation to natural population remains uncertain. Accordingly, no detectable short-term physiological impairment was observed under the tested conditions, although this may not exclude the occurrence of sublethal, chronic, or life-stage-specific effects.

While these results provide an encouraging baseline, several critical limitations must be acknowledged to prevent over-generalization. First, life-stage sensitivity: this study focused on adult mussels, whereas larvae and juveniles typically show higher susceptibility to chemical stressors. Secondly, temporal scope: the experiment focused on acute responses, which may not capture the chronic, cumulative effects of multi-year-infrastructure presence. Thirdly, the absence of biochemical or molecular biomarkers such as oxidative stress enzymes or stress-related gene expression limits the ability to detect early cellular disturbances. Additionally, direct characterization and quantitative measurement of the chemical compounds released from the tested paint were beyond the scope of the present study, as exposure conditions were supported by a parallel investigation. This study design may have constrained the interpretation of the link between the observed physiological responses and specific leachate profile, concentrations, and their spatial and temporal variability.

Collectively, these limitations highlight the need for further research

to develop a comprehensive risk assessment framework for the expanding offshore wind sector in the North Sea. Future studies should (i) conduct medium to long-term exposures (4–10 week) exposure experiments to assess bioaccumulation and chronic toxicity, (ii) incorporate multi-stage life cycle assessments, focussing on pre- and post-settlement phases which are vital for population recruitment on artificial reefs, and (iii) apply advanced molecular techniques to identify sub-lethal metabolic disturbances, alongside direct chemical analysis to of the specific chemical substances and the determination of their concentrations in the experimental environment. Taken together, the current study provides preliminary baseline data relevant to the North Sea's blue growth strategy, supporting informed adaptive management, policy development, and industry practices aimed at balancing renewable energy expansion with the maintenance of marine ecosystem health (Spaniol and Rowland, 2022; van den Burg et al., 2019).

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CRediT authorship contribution statement

Gudrun De Boeck: Writing – review & editing, Validation, Supervision, Methodology, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Lieven Bervoets:** Writing – review & editing, Validation, Supervision, Resources, Project administration, Methodology, Funding acquisition, Conceptualization. **Bavo De Witte:** Writing – review & editing, Validation, Supervision, Resources, Project administration, Methodology, Funding acquisition, Conceptualization. **Moses Ndugwa:** Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

Declaration of Competing Interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Lieven Bervoets reports financial support was provided by Interreg North Sea. If there are other authors, they declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at [doi:10.1016/j.risma.2026.105085](https://doi.org/10.1016/j.risma.2026.105085).

Data Availability

Data will be made available on request.

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