Modeling the impact of salinity variations on aquatic environments: including negative and positive effects in life cycle assessment

Alba Roibás-Rozas[a], Montserrat Núñez[*b], Anuska Mosquera-Corral[a] and Almudena Hospido[a]*

[a] Cross-disciplinary Research Center in Environmental Technologies (CRETUS), Department of Chemical Engineering, Universidade de Santiago de Compostela, 15782 Santiago de Compostela, Galicia, Spain.

[b] Sustainability in Biosystems, Institute of Agrifood Research and Technology (IRTA), 08140 Caldes de Montbuí, Barcelona, Spain.

[*]Corresponding author e-mail: montserrat.nunez@irta.es; Phone: +34934674040.

ABSTRACT

Salinity is changing in aquatic systems due to anthropogenic activities (like irrigation or dam management) and climate change. Although there are studies on the effects of salinity variations on individual species, little is known about the effects on overall ecosystems, these impacts being more uncertain in transitional waters such as estuaries or fiords. The few works that do address this topic have considered these impacts using ecotoxicity models. However, these models state that an increase in the concentration of a pollutant generates an increase in the impacts, disregarding the effects of water freshening. The present research work introduces a general framework to address the impacts of salinity variations, including emission-related positive effects. We validated this framework by applying it to an estuarine area in Galicia (northwestern Spain), where sharp drops in the salt concentration have caused mass mortalities of shellfish in recent decades. This research work addresses for the first time the potential effects on the environment derived from a decrease in the concentration of essential substances, where the effects of an emission can also generate positive impacts. Moreover, it is expected that the framework can also be applied to model environmental impacts of other essential substances in life cycle assessment (LCA), such as metals and macronutrients.

Keywords: Biodiversity, Climate Change, Ecotoxicity, Life Cycle Impact Assessment (LCIA), Salinity, Species Sensitivity Distribution, Transitional Waters

SYNOPSIS

We propose a new method to evaluate the effects of variations, both increases and decreases, in the concentration of essential substances in aquatic ecosystems.

INTRODUCTION

Climate change is altering the biogeochemical cycles on the planet[1] and shifting water temperature, pH, dissolved oxygen concentration, and salinity[2]. In fact, direct relationships between anthropogenic CO_2 release and alterations in the water cycle which result into salinity variations have been already established[3–5]. Other direct human activities, such as irrigation[6,7], industrialization and agricultural expansion[8], effluent disposal[9,10], or dam management[11], are also behind these salinity variations. These changes and their effects can be even more important in transitional waterbodies, such as estuaries, deltas, or coastal lagoons, which constitute less than 5% of the brackish areas worldwide but provide about half of the global fish catch[8].

However, to the best of our knowledge, just five research works have tried to model the impacts of salinity changes on the environment[12]. Although a new impact category addressing salinity was proposed[13–15], to the best of our knowledge, it has never been included in an LCA study[16]. Two assessment methodologies focused on soils, modeling salinity impacts according to variations in soil conductivity and linking them with food production and crop diversity loss[17,18]. Two other methodologies focused on aquatic systems, shaping the effects of salinity variations according to ecotoxicity models for water environments[6,9]. The latter were only partially successful for several reasons, as ecotoxicity is currently based on the observation that the sensitivities of different species to a chemical follow a normal distribution, so increased exposure will generate increased impacts[19]. However, the impact of salinity is not only linked to concentration increases, but also to concentration decreases (systems can become saltier or fresher), so an approach based on ecotoxic models would fail to describe these effects. Moreover, salt is not a pollutant or a toxic, but an essential element, so ecotoxic models might not be valid to describe its behavior. Hence, a critical improvement of these methodologies is necessary. Therefore, the aim of this study is to provide a framework for the evaluation of the impacts of salinity changes in aquatic ecosystems.

Estuaries are important ecosystems from an ecological point of view[11], supporting highly productive communities of arthropods, mollusks, fish, and birds, as well as complex food webs[20]. The Western *rías* of the northwest Spanish region of Galicia, vital for the local economic and social development, are partially mixed estuaries where partial stratification is maintained by the river discharge in the winter and by solar heating in the summer[21]. Due to their uniqueness, it is common that geography experts call these areas by their name in Galician (*rías*)[22]. Nevertheless, in the past few decades, events of massive shellfish deaths were reported in these estuaries due to sharp decreases in the concentration of salt in the *rías*[11,23–25]. These occurrences have been linked to heavy rains, locally managed freshwater releases from river dams, and increased river runoffs, where the frequency of these climatological episodes is supposed to increase in the coming years due to climate change[24]. Among the Galician *rías*, the biggest and most productive one is the *Arousa ría*[11,24,26]. Therefore, this *ría* was chosen to test and validate the methodology proposed in the present research work.

In sum, we describe in this paper a general model shaping the effects of salt variations in aquatic systems. Then, we apply the model to a case study for the procedure validation. By providing this framework, we expect to expand the current knowledge of transitional water systems, allowing to integrate the evaluation of impacts (positive and negative) in environmental sustainability assessments, and to improve, for example, effluent disposal or dam management control in sensitive areas.

MATERIALS AND METHODS

Salinity is defined as the concentration of several inorganic ions (including Na^{+}, Ca^{2+}, Mg^{2+}, K^{+}, Cl^{-}, SO_4^{2-}, CO_3^{2-}, NO_3^{-} and HCO_3^{-})[12,16], where most of these elements are essential substances needed for organisms to live. In the present research work, salinity is referred to as the concentration of NaCl, in kilograms per cubic meter (kg/m^3), as it is abundant in water streams considered saline.

Regarding the methodology, impacts linked to chemical releases are measured as in Eq. (1), where IS is the impact score, CF is the characterization factor, and M the mass of substance emitted. Then, the CF is calculated considering the principal cause-effect chains – thus, through fate, exposure, and effect factors (FF, XF and EF, respectively), according to Eq. (2)[27].

$$\mathrm{IS} = \mathrm{CF}\cdot\mathrm{M} \tag{1}$$

$$\mathrm{CF} = \mathrm{FF}\cdot\mathrm{XF}\cdot\mathrm{EF} \tag{2}$$

CFs addressing impacts on ecosystem quality (aka natural environment) at the endpoint level have units of potentially disappeared fraction of species (PDF)$\cdot m^3\cdot$time/kg. As M in Eq. (1) is in kilograms, IS has units of PDF$\cdot m^3\cdot$time[27]. FF in Eq. (2) is expressed in units of time (it represents the mass of a chemical in the environment resulting after an emission flow, so units are kg/(kg/day), which yields days), XF is dimensionless, and

EF is expressed as PDF·m^3/kg. The XF represents the availability of the released chemical in a system (the fraction of the pollutant that is dissolved in the water), which can be considered as 1 for salt as it is fully dissolved[9].

- **FATE FACTOR (FF)**

The FF is linked to the physical behavior/distribution of the substance in the environment and normally expresses its persistence in units of time[19,27,28]. However, the FF also represents the predicted mass residence of a substance in a receiving compartment per unit of emission flow into it[19], so the FF is calculated by applying mass balances to the studied compartment in which degradation and transfer processes are described[28]. Salt do not degrade (i.e. salt's residence time on the environment equals millions of years[9]), so the mass balances will describe how they are transferred, and the relevant flows entering and leaving an aquatic compartment. For the calculation of an FF for salinity, the streams considered are shown in Figure 1:

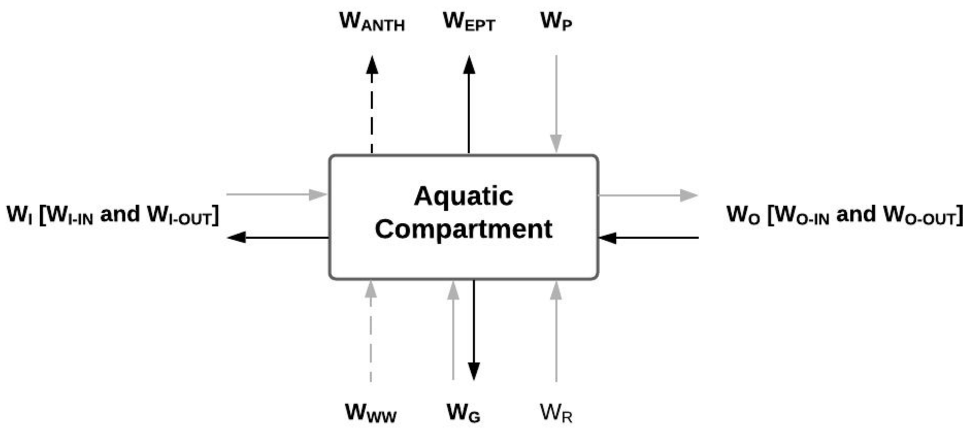


Figure 1. Streams to be considered in the mass balances for quantifying the FF. Gray lines represent the inflows and black lines the outflows. Dashed lines are the anthropogenic flows, and continuous lines are natural streams. Wi refers to each stream, where I = main inlet stream, O = main outlet stream (for both, I/O-OUT is the outflow and I/O-IN is the inflow of the stream). ANTH = anthropogenic outflows, WW = anthropogenic inflows (wastewater), EPT = evapotranspiration, G = groundwater, P = precipitations, and R = runoff.

- W_I refers to the main inlet stream, where returns from the studied compartment can take place. In a marshland, for example, the main inflow ($W_{I\text{-}IN}$) is the river, and the W_I outflow ($W_{I\text{-}OUT}$) represents infiltrations of saline water into the fluvial system.
- W_O streams represent the main outflow from the system ($W_{O\text{-}OUT}$), where the flow can also occur in both directions; thus, returns from the main outflow to the

124 compartment ($W_{O\text{-}IN}$) can occur. For example, for an open estuary, these flows represent

125 the tidal, so this main outstream would be subjected to bidirectional flows.

126 - W_P, W_{EPT}, W_G, and W_R are climate and water cycle-related streams that

127 represent the precipitation, the evaporation, the groundwater, and the runoff of the

128 system, respectively. Note that the groundwater flow could also be bidirectional, as it

129 can feed the compartment, but infiltrations from the system to the groundwater network

130 can also occur.

131 - W_{ANTH} and W_{WW} are the anthropogenic outflows and inflows in the system,

132 respectively. W_{ANTH} represents the water taken from the compartment for anthropogenic

133 purposes (for example, for irrigation), and W_{WW} is the poured water into the system

134 (normally, treated wastewater).

135 Once the streams are identified and quantified, the salt concentration of each flow (S_i,

136 kg NaCl /m^3) is also needed for the calculation of the FF (Eq. (3), in units of time). $\overline{S}$ is

137 the average salt concentration in the system (kg/m^3), $\overline{V}$ the average volume of the

138 studied compartment (m^3), and $\overline{F_i}$ the average mass flow of the calculated F_{IN} and F_{OUT}

139 (kg salt/time) (Eq. (4)), which represent the total mass of salt in the inlet and outlet

140 flows, respectively. F_{IN} should equal F_{OUT} as no accumulation takes place and the model

is then valid for steady state, but the FF is quantified using $\overline{F_i}$ (an average) to acknowledge possible discrepancies of experimental data. Additionally, $\overline{F_i}$ can be calculated yearly, monthly, or seasonally, defining the units of the factor itself.

$$FF = \frac{\bar{S}\cdot\bar{V}}{\overline{F_i}} \tag{3}$$

$$\overline{F_i} = \sum W_{i-IN}\cdot S_{i-IN} = \sum W_{i-OUT}\cdot S_{i-OUT} = F_{IN} = F_{OUT} \tag{4}$$

- **EFFECT FACTOR (EF)**

The EF quantifies the fraction of living species that are going to potentially disappear in the aquatic ecosystem by the release of a certain chemical[19,27,28]. The available literature regarding salinity variation effects in aquatic ecosystems models these impacts according to ecotoxic methodologies, such as USEtox[6,9]. Under that approach, the EF is calculated employing a species-sensitivity distribution (SSD) curve which represents the sensitivity of an entire ecosystem to a substance[27]. However, it is constructed based on the premise that increased exposures will lead to increased effects[19]. Indeed, if the salinity of a fresh aquatic environment increases, the organisms living in the system will be negatively affected[10,29–32]. However, negative effects in aquatic ecosystems are also

155 reported when salinity decreases, such as massive mortalities reported after exposure to
156 low salt concentrations due to climatologic events or anthropogenic actions[11,33,34].
157 Moreover, some limitations of SSD curves to model ecotoxic impacts due to salinity
158 have been pointed out[35], and other indicators, such as species richness, have been
159 proposed for studies assessing the effects of salinity[31].

160 Therefore, variations in the concentration of essential substances can provoke effects in
161 several directions, as an increase in the salt concentration can be potentially beneficial
162 for an ecosystem and vice versa, meaning that emission-related impacts can be positive
163 in some cases. Moreover, currently, it is considered that a substance provokes toxic
164 effects if it enters in an organism and causes poisoning, endocrine disruption, or other
165 lethal effects[19]. This approach might be partially accurate for effects of exposure to high
166 salt concentrations, but it fails when defining the observed effects on ecosystems for
167 decreases in salinity. This is the reason why the current applied approach (using
168 ecotoxic methodologies) to model the impacts due to variations in essential substances
169 (such as salts) need to be reconsidered.

170 EFFECT FACTOR FOR SALINITY VARIATIONS IN AQUATIC ENVIRONMENTS

The proposed approach is based on the premise that the species in an ecosystem have an optimal range of salinity for living, and that detrimental effects will be observed if it varies below or above it[36]. Figure 2 shows an ideal representation of a transitional water body, where there is an optimal concentration of salt at which no impact occurs (i.e., PDF = 0). Please note that this figure represents a theoretical hypothesis to be later verified by the case study application. Then, negative effects will occur if salinity increases above the optimal range or decreases below it. Moreover, positive impacts will occur for increments in the salt concentration for environmental concentrations below the optimal, and vice versa. Note that, for salinity increases in the high range, the function follows a distribution like the classic SSD curve.

The first step is then to define the optimal salt concentration range by gathering data of chronic effects for the species of the ecosystem regarding salinity (also needed to quantify the EF itself). If the data are expressed as acute, an acute-to-chronic ratio (ACR), which is generally 2[28], can be used by dividing or multiplying the salt concentration by the ACR in the high range or in the low range, respectively. Then, according to our hypothesis (effects will happen above and below a favorable range), that data representing the chronic effects linked to salinity will be shaped like in the examples provided in Figures 2a, 2b, or 2c.

For water systems from oligohaline to hyperhaline (see Table S1 at the supporting information (SI)), it is expected that the concentration-to-response curves have a shape like the ones shown in Figure 2a or 2b, where the optimal (environmental) salt concentration can be estimated as a range (the points for which the function has values $y < 0$, Figure 2a) or a point (the minimum value of y, Figure 2b). For freshwater environments, it is more likely that the impacts in the ecosystem are due to salt concentration increases. For the part of the curve representing salinization, a classic SSD approach might be a fair approximation. Nevertheless, the whole system profile is only provided if both ranges (high and low) are included, where the optimal (environmental) concentration can be found as the intersection of the two functions (a lineal one for the low range and SSD-like for the high range, Figure 2c).

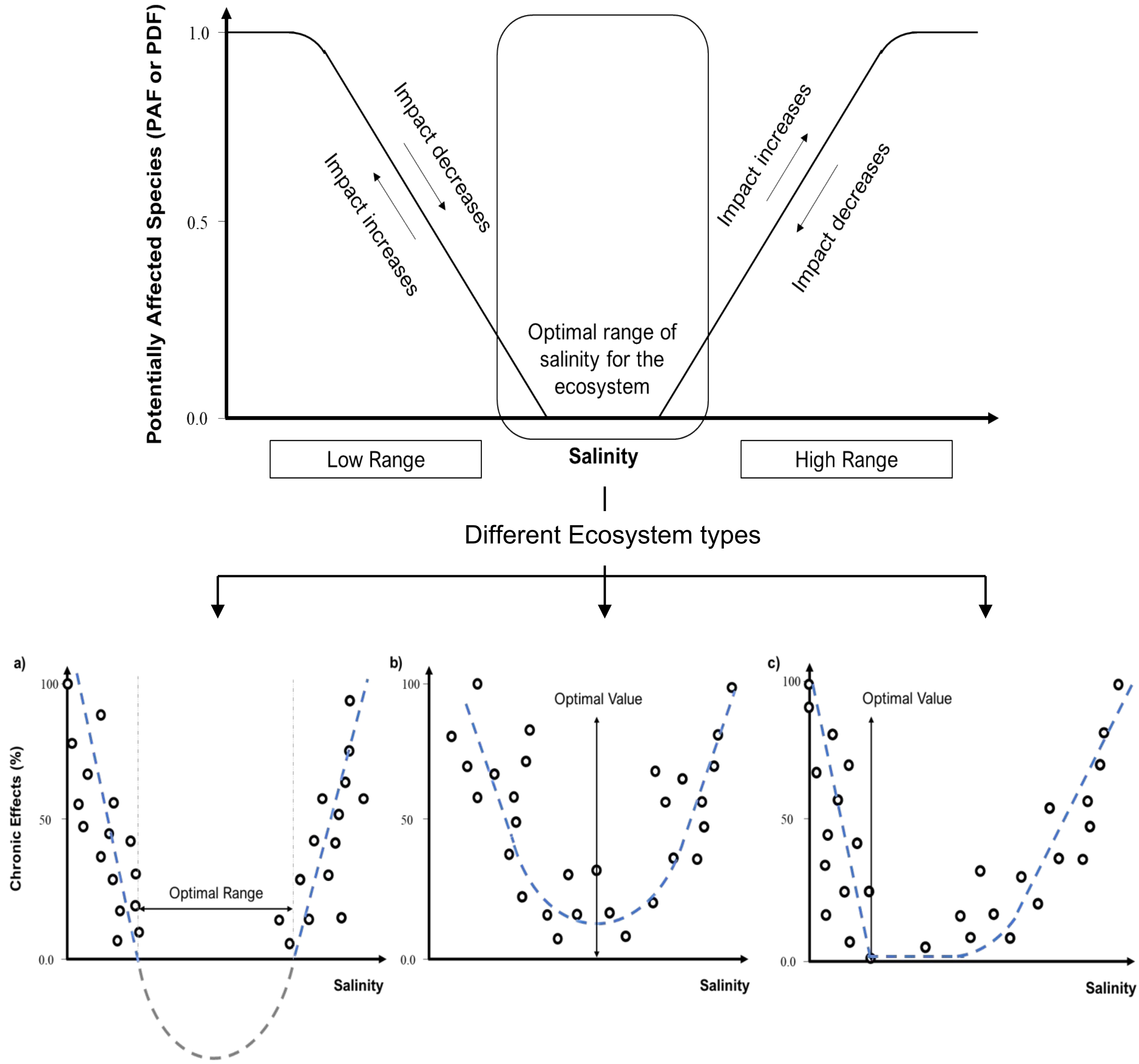


Figure 2. Expected (hypothetical) shape of the curve defining the optimal salinity range for aquatic ecosystems. The main plot is an ideal representation of the impacts of salt variations, and the graphs a, b, and c are theoretical representations of different types of ecosystems, where a and b shape the impacts of transitional or non-fresh compartments, and c is a freshwater system.

After defining the optimal region, there are two EFs: EF_{LOW} and EF_{HIGH}, both in PDF·m^3/kg. They represent the amount of a substance that generates a certain effect on the ecosystem, where the EF is the slope of the concentration-response curve. $HC50_{LOW}$ and $HC50_{HIGH}$ (both in kg NaCl/m^3) represent the salt concentration that generates an effect in 50% of the ecosystem species in the low and high range, respectively; thus, for effects in 50% of the species, EFs are expressed as in Eq. (5) and Eq. (6). HC50s are the geometric mean of the individual species EC50[28], which is the concentration of pollutant that generates an effect in the 50% of the individuals of a single species. In this sense, an SSD-like approach is still maintained.

$$EF_{LOW} = \frac{0.5}{HC50_{LOW}} \tag{5}$$

$$EF_{HIGH} = \frac{0.5}{HC50_{HIGH}} \tag{6}$$

Assuming a classic SSD-like curve (i.e., log-logistic functions), EC50s could be calculated using to mirrored sigmoidal curves. Moreover, according to our hypothesis (Figure 2), EC50s could be estimated by fitting the chronic effect data to a quadratic function (see Figure S1 at the SI). Note that the log-logistic and the quadratic curves only converge at intermediate ranges of effects, so this approximation might not be accurate to calculate EFs based on HC10 or HC20, as new trends on ecotoxicity modeling suggest the use of EC10s or EC20s to model the factors[19]. Finally, the sign of each EF will be established depending on the direction of the change: increasing the salt concentration in the low range accounts for a negative EF_{LOW} (impacts decline), and decreasing it accounts for a positive EF_{LOW} (impacts increase), and vice versa for the high range.

APPLICATION OF THE METHODOLOGY TO A CASE STUDY

- FATE FACTOR CALCULATION

For the case of *Arousa ría*, Figure 1 is simplified: W_G is neglected due to the insignificant contribution of the groundwater to the total flow of the *ría*[37], and $W_{I\text{-}OUT}$ is ignored since no infiltrations from the estuary to the river are expected (see Figure S2). The remaining streams are quantified in cubic meters per month, covering from 2011 to 2018 according to the data availability, as detailed bellow. First, stream flows are determined:

237 - $W_{I\text{-}IN}$ corresponds to the flows of rivers Ulla and Umia, which are monthly measured and reported by local administrations[38] in several locations along the river courses, so the closest point to the *ría* was chosen.

240 - W_{WW} can be divided into fresh wastewater ($W_{WW\text{-}FRESH}$) and salty one ($W_{WW\text{-}SALT}$). It was estimated that the area of *Arousa ría* is responsible for about 6.6% of the total (W_{WW}) Galician discharges[37,39] and that 82–92% of this amount (yearly variation in 2011-2018) corresponded to fish-canning wastewater[37,39], the latter representing virtually all saline emitting sectors ($W_{WW\text{-}SALT}$, in cubic meters per year). Then, to obtain monthly data, a distribution was defined based on the shellfish and seafood harvesting and processing seasonal pattern to estimate $W_{WW\text{-}SALT}$ in m^3/month (see Table S2). On the other hand, $W_{WW\text{-}FRESH}$ (including urban wastewater and fresh industrial streams, in cubic meters per year) was estimated as the subtraction of $W_{WW\text{-}SALT}$ from the total W_{WW}. Then, to obtain monthly data, it was assumed that fresh discharges are evenly distributed along the months of the year (also considering that freshwater streams represent a flow considerably lower than salty streams).

252 - W_{ANTH} is not linked here to irrigation, as estuary water has high salinity and Galicia is an area with high precipitations, but the local fish-canning industries use

seawater for their processes with variable flows. Due to the lack of more precise data, we assumed that 75% of water input for fish-canning processes was saline water and 25% was freshwater[40–43] and that there are no water losses in the process, i.e., inputs = outputs, thus $W_{ANTH} = 0.75·W_{WW\text{-}SALT}$· .

- Evaporation rates were estimated using empirical correlations as indicated in Eq. (7)[44,45], where W^*_{EPT} is stated in L/m^2·time (i.e., monthly averages in this case), u_2 is the wind speed (m/s), A is the evaporating area (m^2), v^*_w is the saturated vapor pressure at the water surface temperature (kPa), and v_a is the partial vapor pressure in the air at 2 m height (kPa) calculated by Eq. (8), where v^*_a is the saturated vapor pressure at air temperature (kPa) and ϕ is the relative humidity (%). Finally, to obtain W_{EPT} in equivalent units to the rest of the streams (m^3/month), Eq. (9) is needed.

$$W^*{}_{EPT} = (2.36 + 1.72·u_2)·A^{-0.05}·(v_w^* - v_a) \tag{7}$$

$$v_a = v_a^*·\phi \tag{8}$$

$$W_{EPT} = \frac{W_{EPT}^*·A}{1000} \tag{9}$$

- Average monthly precipitations (W_P) were obtained from the meteorological station located in Corón (Vilanova de Arousa)[46].

- W_R (runoff) was taken as the average values of the South West basins of Galicia[47]. Being the only data set available, the monthly distribution and expected runoff flow for 2002 were applied to the 2011-2018 period defined.

At this point, all the flows, except $W_{O\text{-}IN}$ and $W_{O\text{-}OUT}$, which represent the tidal, are quantified, so mass balances must be solved. First, Eq. (10) and Eq. (11) present the general steady state water and the salt balances, respectively, where S_i is the salt concentration (kg NaCl/m^3) of each stream W_i.

$$W_{O-IN} + W_I + W_R + W_P + W_{WW} = W_{O-OUT} + W_{EPT} + W_{ANTH} \qquad (10)$$

$$W_{O-IN} \cdot S_{O-IN} + W_I \cdot S_I + W_R \cdot S_R + W_P \cdot S_P + W_{WW} \cdot S_{WW} = W_{O-OUT} \cdot S_{O-OUT} + W_{ANTH} \cdot S_{ANTH} \qquad (11)$$

Note that Eq. (10) could be written as $W_{IN} = W_{OUT}$, representing the total inflows and outflows in the estuary. Then, to accurately quantify the tidal streams, new mass balances will be stated, representing a system as natural as possible (Eq. (12) and Eq. (13)), where no anthropogenic activity exists:

$$W_{O-IN} + W_I + W_R + W_P = W_{O-OUT} + W_{EPT} \qquad (12)$$

$$W_{O-IN} \cdot S_{O-IN} + W_I \cdot S_I + W_R \cdot S_R + W_P \cdot S_P = W_{O-OUT} \cdot S_{O-OUT} \qquad (13)$$

To be able to proceed, the salt concentration of each stream, S_i, needs now to be quantified:

- The salt concentration of the ocean (S_{O-IN}) was considered constant at 36 g NaCl/L, as it is the average estimated sea surface salinity measured between 2004 and 2013 for this Atlantic area by satellite monitoring[4].
- The salt concentration of the estuary (S_{O-OUT}, S_{ANTH}, but also $\bar{S}$ in Eq. (3)) is variable according to the spatial distribution and the tidal intensity. Water is fresher near the river mouth and saltier near the ocean, but the salt concentration also increases with depth because salty water has a higher density[48]. Data collected from two buoys (*Ribeira*, in an intermediate location between the river mouth and the ocean, and *Cortegada*, near the river mouth; see map in Figure S3) [46] were used to determine the monthly average salinity of the estuary.
- Galician rivers are not salinized, presenting low salinity and conductivity[49]. Normally, chloride concentrations of Spanish rivers range between 0.010 and 0.030 kg Cl^-/m^3, and recent studies for Galician rivers near shore areas reported values of 0.0103 – 0.0222 kg Cl^-/m^3 [50]. An average value of 0.020 kg $NaCl/m^3$ is considered for S_I and S_R.

295 - Chloride concentration in rainwater is generally very low, although it varies
296 worldwide depending on wind intensity and seawater proximity. Measurements of the
297 chloride concentration in rainwater in Spain and Portugal averaged 0.020 kg Cl^-/m^3, and
298 the value reported for the Galician station, located at around 40 km from the coast, was
299 0.003 kg Cl^-/m^3 [51]. A slightly greater salinity of 0.005 kg $NaCl/m^3$ is applied for S_P to
300 consider the effect of sea salt aerosols.

301 - Eq. (14) and Eq. (15) are used to calculate S_{WW}, where $W_{WW\text{-}SALT}$ is the yearly
302 flow of industrial fish canning wastewater discharged into the estuary (m^3/year), and $\overline{S}_t$
303 is the average salt concentration of the estuary each year (kg/m^3). It was assumed that
304 $S_{WW\text{-}FRESH}$ coming from drinkable sources was 0.5 kg $NaCl/m^3$ [52], and that $S_{WW\text{-}SALT}$
305 contained 75% of estuarine water and 25% of fresh (drinkable) water as previously
306 indicated. The salinity of waste streams poured into the *ría* S_{WW} can be estimated as an
307 average value as shown in Eq. (15).

$$S_{WW-SALT} = \frac{0.75 \cdot W_{WW-SALT} \cdot \overline{S}_t + 0.25 \cdot W_{WW-FRESH} \cdot 0.5}{W_{WW-SALT}} \tag{14}$$

$$S_{WW} = \frac{W_{WW-SALT} \cdot S_{WW-SALT} + W_{WW-FRESH} \cdot S_{WW-FRESH}}{W_{WW-SALT} + W_{WW-FRESH}} \tag{15}$$

Finally, the FF can be calculated. As already stated, both monthly and yearly results can be obtained, as well as seasonal values, the dry season spanning June, July and August in the case study area, and the remaining months are the wet season. To see detailed monthly/seasonal/yearly results and other detailed information about the calculation procedures, see the Excel file and the Section SIII in the SI (see Table S3).

- EFFECT FACTOR CALCULATION

To calculate the EFs, data of chronic effects were gathered (for this study, only mortality was considered). Then, the collected data were represented together to find the ecosystem optimal (environmental) salt concentration (Figure 3). Note that this value is not going to be employed in any other further calculation, and it is just used to determine which data will be utilized to generate EF_{LOW} and EF_{HIGH}. In this sense, several fittings could be tested, as shown in Figure 2a and 2b. Different approaches were applied, evaluated, and discussed (see Section SIV of the SI, and Figures from S4 to S7) to identify this cutoff point. These strategies pointed out a possible environmental optimal range of 24–36 g NaCl/L, where the average point obtained after testing different fittings is 31.9 ± 1.4, which will be the cutoff to define the low and high ranges of salt concentration comprising EF_{LOW} and EF_{HIGH}, respectively.

Therefore, EC50 concentrations are calculated for each species at high and low range considering the estimated cutoff point. Most of the information gathered referred to effects at low range, so the $EC50_{LOW}$ for these species can be obtained directly using a sigmoidal curve with a profile mirrored to the classic SSD curve fittings. However, $EC50_{LOW}$ and $EC50_{HIGH}$ can be also estimated for each species by using the quadratic approach of our hypothesis, which provides effects at both ranges (as in Figure S1; see Table S4 for detailed information of how the quadratic approximation was used).

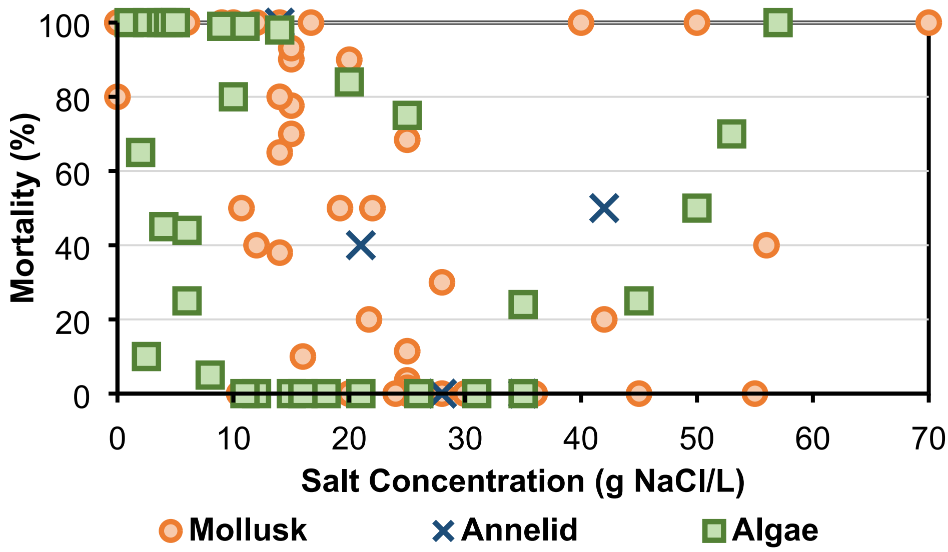


Figure 3. Distribution of the data regarding mortality of different species present in *Arousa ría*. The data used to generate this plot are specified in Table 1 and Table S4 and Section SIV of the SI.

The value of each EC50 (high and low) and both HC50s are shown in Table 1. The numbers in italic indicate that the concentration was obtained by the quadratic fitting using data from the low range of salinity (for *V. senegalensis* and *S. polyschides*, the power function had a convex profile, so the data were estimated using the inverse function, see Table S4). The results of EF_{HIGH} and EF_{LOW} (Table 1) fit with the expected system behavior, as biotic diversity in estuaries starts to decline above a salinity of about 40 kg NaCl/m^3, with most species unable to survive in salinities above 50 kg NaCl/m^3 [36].

- CHARACTERIZATION FACTOR CALCULATION

CFs were obtained according to Eq. (2). Here, the EF achieves different values according to the level of salinity (high and low), while the FF, which is supposed to be the same for both ranges of salinity (the water streams will have the same physical distribution regardless of the salt concentration), varies seasonally (wet/dry season). Therefore, there are six possible CFs to use (Table 1).

Table 1. Fate Factors, Effect Factors and Characterization factors modeling the effects of salinity variations in *Arousa ría*. For the EF section, the EC50s in italic represent the values estimated assuming quadratic distribution (data for chronic effects were only

available for low range). The results are expressed as average ± standard deviation and the confidence interval are between brackets, where negative results were assumed zero.

<table>
<tr><th colspan="6">Fate Factors (month/year)</th></tr>
<tr><th colspan="2">Wet (month)</th><th colspan="2">Dry (month)</th><th colspan="2">Annual (year)</th></tr>
<tr><td colspan="2">6.84 ± 1.84</td><td colspan="2">4.02 ± 1.52</td><td colspan="2">4.51 ± 1.51</td></tr>
<tr><td colspan="2">[1.32, 12.36]</td><td colspan="2">[0, 8.59]</td><td colspan="2">[0, 9.05]</td></tr>
<tr><th colspan="6">Effect Factors, HC50s and EC50s</th></tr>
<tr><th colspan="4">Species</th><th>$EC50_{LOW}$</th><th>$EC50_{HIGH}$</th></tr>
<tr><td colspan="4">Ruditapes philipinarium (Japanese clam)[11,53]</td><td>16.0</td><td>34.4</td></tr>
<tr><td colspan="4">Ruditapes decussatus (Grooved carpet shell)[11,54]</td><td>15.5</td><td>42.9</td></tr>
<tr><td colspan="4">Cerastoderma edule (Common cockle)[11,55]</td><td>20.4</td><td>61.7</td></tr>
<tr><td colspan="4">Vanerupis corrugata (Pullet carpet shell)[11,53]</td><td>21.7</td><td>37.8</td></tr>
<tr><td colspan="4">Saccharina latissima (Sea belt, brown algae)[56,57]</td><td>8.6</td><td>39.1</td></tr>
<tr><td colspan="4">Diopatra neapolitana (Polychaete)[58]</td><td>19.0</td><td>42.3</td></tr>
<tr><td colspan="4">Donax trunculus (Wedge clam)[59]</td><td>19.3</td><td>32.1</td></tr>
<tr><td colspan="4">Mytilus galloprovincialis (Mediterranean Mussel)[60]</td><td>9.8</td><td>38.2</td></tr>
<tr><td colspan="4">Scrobicularia plana (Peppery furrow shell)[55]</td><td>10.7</td><td>40.4</td></tr>
<tr><td colspan="4">Saccorhiza polyschides (Furbellow, brown algae)[61]</td><td>30.1</td><td>40.1</td></tr>
<tr><td colspan="4">Zostera noltei (Dwarf eelgrass, seagrass)[62,63]</td><td>1.3</td><td>49.6</td></tr>
<tr><td colspan="4">HC50 (Geometric mean)</td><td>12.7</td><td>41.1</td></tr>
<tr><td colspan="4">HC50 arithmetic mean</td><td>15.7 ± 7.8</td><td>41.7 ± 8.0</td></tr>
<tr><th colspan="6">Effect Factors (PDF·m³/kg)</th></tr>
<tr><th colspan="3">EF_{LOW}</th><th colspan="3">EF_{HIGH}</th></tr>
<tr><td colspan="3">0.04 ± 0.02</td><td colspan="3">0.01 ± 0.002</td></tr>
<tr><td colspan="3">[0, 0.1]</td><td colspan="3">[0.005, 0.02]</td></tr>
<tr><th colspan="6">Characterization Factors (PDF·month·m³/kg)</th></tr>
<tr><th colspan="3">CF_{LOW}</th><th colspan="3">CF_{HIGH}</th></tr>
<tr><td>Dry</td><td>Wet</td><td>Annual</td><td>Dry</td><td>Wet</td><td>Annual</td></tr>
<tr><td>0.27 ± 0.21</td><td>0.16 ± 0.14</td><td>0.18 ± 0.15</td><td>0.08 ± 0.04</td><td>0.05 ± 0.03</td><td>0.05 ± 0.03</td></tr>
<tr><td>[0, 0.89]</td><td>[0, 0.57]</td><td>[0, 0.62]</td><td>[0, 0.20]</td><td>[0, 0.13]</td><td>[0, 0.14]</td></tr>
</table>

For the obtained FFs, the standard deviation acknowledges the discrepancy of steady state assumption (see Section SIII of the SI). For the EF, the confidence intervals and the standard deviation of the factor were calculated by using the arithmetic average (see Table 1). For all factors, the uncertainty intervals were obtained considering three times the standard deviation, and negative lower bounds were considered zero.

CFs are reported as absolute values, but they depend on the direction of the change (i.e., increase or decrease in salt concentration at low or high range). Therefore, to quantify the effects of a dam release during the wet season which provokes a decrease in the salt concentration, CF = 0.16 PDF·month·m^3/kg (i.e. impact increases); but to quantify the impact of a saline effluent discharge in the same situation (wet season, low salinity), CF = -0.16 PDF·month·m^3/kg should be used as salinity is already below the optimal, so the increase in salt concentration has a positive effect in the receiving waters. In this example, CF_{LOW} is used instead of CF_{HIGH} because *Arousa ría* has a salt concentration below the optimal[49].

DISCUSSION

When evaluating the effects of pollutants' release, it is logic to assume that a rise in the chemical concentration generates an increase in the impacts. However, anthropogenic activities are affecting the planetary biogeochemical cycles, and impacts are now not only due to the release of harmful substances, but also to variations in the environmental conditions of ecosystems provoked by unnatural changes. There are some substances (e.g., salt, nitrogen, dissolved oxygen) to which the classic ecotoxic impact approach might just fit partially. Indeed, a drastic increase in the substance concentration will generate adverse impacts. However, at the same time, a certain concentration in the media is needed to support the ecosystem survival.

In recent years, concern for salt variations in the different environmental compartments has increased, but the models available to describe their impacts are still scarce and the current literature uses ecotoxicity models to assess these effects[6,12,18,64]. For instance, the USEtox model, one of the most used methods to evaluate ecotoxic impacts in LCA, has not addressed effects of salt releases so far[12] and does not include the coastal seawater and brackish areas as environmental compartments[19]. Therefore, new approaches are necessary to include essential substances and these critical areas in the impact assessment models, and specific methodological choices must be implemented to develop CFs for essential elements in non-freshwater environments.

Regarding the FF, it is expressed in this work in units of time (see the fate factor section in materials and methods). This provides a suitable framework to model the fate of essential substances. The few research works that have assessed the effects of salinity variations in aquatic environments solved the FF question from different approaches. In a first study, when evaluating the impacts of brine disposal[9], the different elements present in brine were grouped. NaCl had a high residence time, so the FF was chosen as the residence time of the second most persistent element of the salinity group (Cu^{2+}, 37 days). However, this approximation can have high errors due to the difference in the elements' residence times (from days to millions of years). In another study, when steady-state mass balances were applied to the water streams and the salt in a coastal wetland[6], the units of the FF were g·year/L, yielding in a dimension of time per concentration. This factor was useful to directly link salt variations in the wetland not only with ecotoxic impacts, but also with social and economic effects as crop loss. However, the unit of that FF complicates the comparison of the results obtained there with the present study. Moreover, it hinders the evaluation of the impacts of salinity variations on the ecosystem linked to different impact categories as ecotoxicity. Therefore, to remain coherent and consistent with other impact categories, it is necessary to provide factors that align with the standard units of the impact assessment stage and that are used by other methodologies recommended by the UNEP's Life Cycle Initiative, such as the USEtox. This harmonization aspect also applies to the

408 measure of the EF, which is PDF in this study because UNEP's Life Cycle Initiative
409 recommends to base ecosystem damage estimates on this metric[19,65].

410 As for the EF, the proposed approach copes with the fact that impacts might be due to
411 increases but also decreases in the concentration of essential substances, and it
412 acknowledges potential benefits linked to emission-related impacts. In this sense, a
413 classic SSD-based methodology (typically used in ecotoxicity) appears as a practical
414 and useful tool to predict the possible negative effects of a pollutant's release on
415 ecosystems. However, although some of the limitations linked to the use of SSD curves
416 have been pointed out in the past few years, the principles shaping the methodology
417 have remained unchanged for decades[35]. Additionally, the use of classic ecotoxicity
418 methodologies also have some limitations to model impacts due to variations in the
419 concentration of essential substances, since the effects of these variations do not fit the
420 classic definition of toxicity (linked to poisoning, endocrine disruption, etc.).

421 For the present study, a quadratic function was used to shape the effects of salinity on
422 ecosystems instead of the typical log-normal or log-logistic distribution applied in the
423 SSD-based methods, by applying this approximation for the calculation of EC50s (and,
424 therefore, the EFs). Nevertheless, log-logistic distributions were also applied to the

chronic data gathered for each individual species to test the robustness of the approximation. The differences between the EC50s found applying log-logistic and quadratic fittings averaged 3.82% (data not shown), meaning that the quadratic approach is accurate to estimate effects at intermediate ranges of concentration in an LCA context (but might not be accurate for EC10s or EC20s). This approximation was used to estimate the EC50s and HC50s at both concentration ranges (low and high), but the SSD-based methodology fundamentals are maintained. Thus, the current study does not aim to question the SSD curves themselves, but to expand their scope to understand new ecological features.

In fact, this quadratic approximation can be especially useful for systems from oligohaline to hyperhaline (see Table S1), but it might not be as accurate for freshwater systems. Here, the expected effects are linked to salinization, while impacts linked to a sharp freshening are unlikely. In fact, the most important impacts derived from anthropogenic activities and climate change in freshwater ecosystems are expected to be linked to salinization[16], where the SSD-based methodology can be a fair approximation to determine the effects of salinity increase if the effects of freshening are neglected.

The truth is that the few studies considering the effects of salt variations in aquatic environments considered SSD approaches. In a wetland-related study[6], an SSD curve was constructed considering that anthropogenic activities (irrigation) were provoking salinization, which seems to be a fair approximation considering that the salt concentration in the water body had increased from 2.6 g/L in 1983 to 7.50 g/L in 2008. However, the salinity conditions of the wetland were oligohaline, so impacts related to freshening could also take place (due to rainfalls, for example). Therefore, the approximation was fair to evaluate salinization, but failed at evaluating the effects due to potential freshening.

In a brine disposal research work[9], a concentration of 40 kg NaCl/m^3 was chosen as EC50. Analogously to the wetland case, this approximation might be fair considering that desalination plants would discharge their briny effluents in marine waters, and the effects of this saline disposal are expected to be negative impacts linked to salinity increase. However, the modeling of the system is still not fully comprehensive if the low ranges of salinity are no included. In fact, a case where a transitional brackish ecosystem is subjected to freshening and where brine disposal is potentially beneficial could take place under this point of view (impacts would decrease due to brine disposal), so an approach where this is not considered is lacking some relevant information for the system.

Finally, these divergent approaches hinder a discussion of the obtained CF. The wetland and brine CFs, in terms of potentially affected fraction of species (PAF), were 0.32 PAF·yr and 0.47 PAF·m^3·day/kg, respectively, which are similar to the ranges obtained here (0.05 – 0.27 PDF·m^3·day/kg). However, for the wetland, the units are not comparable, and, for both cases, the obtained CFs were at the midpoint level (measuring PAF, not PDF), so the discussion is not straightforward. Nevertheless, it is important to point out how the quantification of CFs to measure the effects of salinity variations can be relevant for the management of anthropogenic activities in sensitive ecological areas, such as transitional waters. In fact, the method described and applied here has the potential to support decision-making processes around effluent discharge, industrial stream management, brine disposal control, and dam flow regulation, by providing useful information about when and how to discharge these anthropogenic streams with s minimum or even a positive impact.

To apply the CFs developed here, the life cycle inventory (LCI) shall record the mass of salts released to the aquatic environment per functional unit, acknowledging possible differences between the wet and the dry season when relevant and if information is available.

FUTURE OUTLOOK

The presented novel approach proposes a model to shape the effects of variations in the concentration of an essential substance in the environment (Figure 4). Here, the effects in the ecosystem might not be always directly proportional to the pollutant concentration. Although the methodology was applied to salt (i.e., NaCl), the same principles can be implemented for other essential substances such as macronutrients, other salts, metals, and even resources such as water. Moreover, as the definition of salinity is broader than just sodium chloride, a comprehensive salinity assessment may shape the effects of varying the concentration of other substances. Furthermore, by including elements such as carbonate, nitrate and sulfate, a comprehensive study of the ecosystem salinity might extend the effects of the observed variations to other environmental categories (such as climate change, eutrophication, or acidification, respectively). In any case, the uniformization of the CFs for salinity impacts (expressed in the consensus units) opens a new pathway where the effects due to variations in the concentration of essential substances can be fully assessed for the first time.

This pathway has a clear bottleneck, which might hinder its extensive application, and which is not new to the LCA community, i.e., regarding data availability. Fortunately, the access to information is becoming easier as science is becoming more accessible and data is being acquired through more sophisticated means, such as dedicated satellites, which can be more easily managed through advanced computational methods. In this sense, although the provided CF only has local applications, the methodological approach is transferable to any other region. Although some data might be difficult to find, a preliminary guidance for data acquirement in the CF development and application is provided in Section SV of the SI.

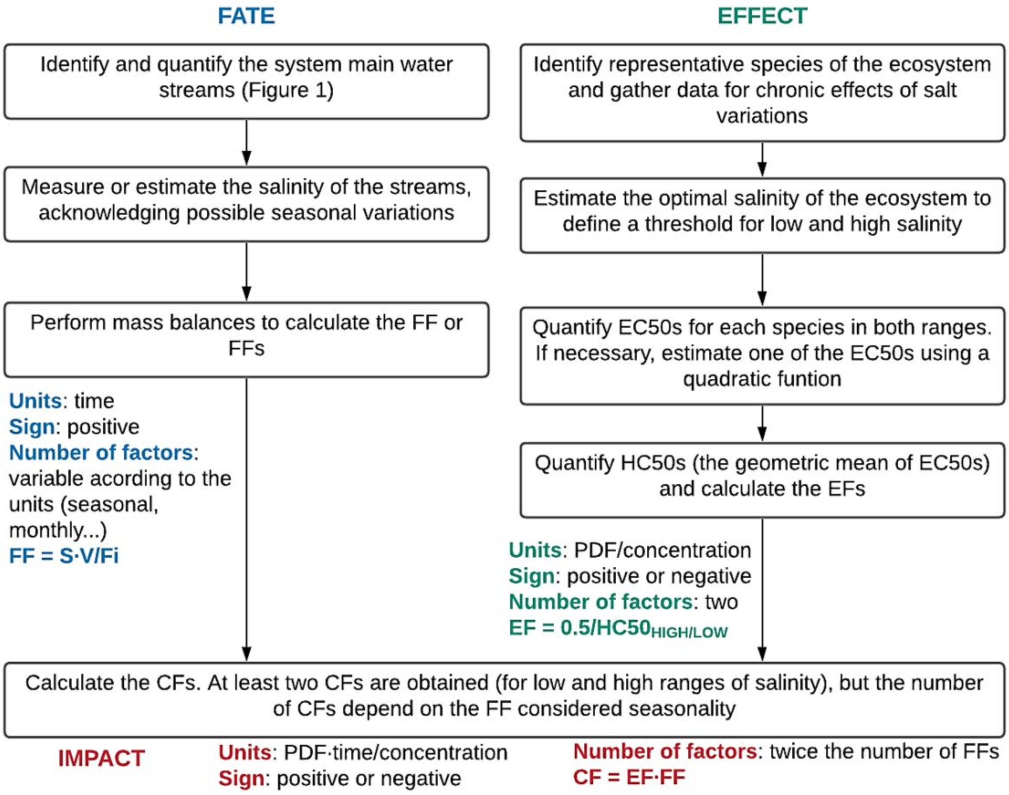


Figure 4. Overview of the steps that must be followed to apply the methodology to assess the effects of salinity variations in aquatic environments.

SUPPORTING INFORMATION

The supporting Word file contains additional information about aquatic environments regarding salinity, about the application of the quadratic approximation for the EF, and about the calculation of the FF. The supporting Excel file contains all the calculations performed to quantify the CF.

ACKNOWLEDGEMENTS

We want to thank Carola Rodríguez Gómez, Victoriano Urgorri and Ana Basáñez Mercader from the Universidade de Santiago de Compostela, Marta Seoane from the Universidade de A Coruña, Jesús de Souza Troncoso and Rula Domínguez from the Universidade de Vigo, Pedro Montero from INTECMAR, and Jorge Diogene and Margarita Fernández from IRTA, for their time and knowledge guiding us on the unfamiliar areas we touched on this research work.

This research was supported by the Spanish Government (AEI) through TREASURE [CTQ2017-83225-C2-1-R] project. A. Roibás-Rozas, A. Mosquera and A. Hospido belong to a Galician Competitive Research Group (GRC), a programme co-funded by the FEDER (EU). M. Nuñez acknowledges the Beatriu de Pinós postdoctoral programme of the Secretary of Universities and Research (Government of Catalonia), the CERCA Programme of the Generalitat de Catalonia, and the Consolidated Research Group TERRA (AGAUR, ref 2017 SGR 1290).

REFERENCES

(1) Breitburg, D.; Levin, L. A.; Oschlies, A.; Grégoire, M.; Chavez, F. P.; Conley, D. J.; Garçon, V.; Gilbert, D.; Gutiérrez, D.; Isensee, K.; Jacinto, G. S.; Limburg, K. E.; Montes, I.; Naqvi, S. W. A.; Pitcher, G. C.; Rabalais, N. N.; Roman, M. R.; Rose, K. A.; Seibel, B. A.; Telszewski, M.; Yasuhara, M.; Zhang, J. Declining Oxygen in the Global Ocean and Coastal Waters. *Science (80-.).* **2018**, *359*

(6371). https://doi.org/10.1126/science.aam7240.

(2) Bramwell, G.; Schultz, A.; Sherman, C.; Giraudeau, M.; Thomas, F.; Ujvari, B.; Dujon, A. M. A Review of the Potential Effects of Climate Change on Disseminated Neoplasia with an Emphasis on Efficient Detection in Marine Bivalve Populations. *Sci. Total Environ.* **2021**, *775*, 145134. https://doi.org/10.1016/j.scitotenv.2021.145134.

(3) Helm, K. P.; Bindoff, N. L.; Church, J. A. Changes in the Global Hydrological-Cycle Inferred from Ocean Salinity. *Geophys. Res. Lett.* **2010**, *37* (18), 2–6. https://doi.org/10.1029/2010GL044222.

(4) Durack, P. J. Ocean Salinity and the Global Water Cycle. *Oceanography* **2015**, *28* (1), 20–31. https://doi.org/10.5670/oceanog.2015.03.

(5) Sun, Q.; Du, Y.; Xie, S.-P.; Zhang, Y.; Wang, M.; Kosaka, Y. Sea Surface Salinity Change since 1950: Internal Variability versus Anthropogenic Forcing. *J. Clim.* **2021**, *34* (4), 1305–1319. https://doi.org/10.1175/JCLI-D-20-0331.1.

(6) Amores, M. J.; Verones, F.; Raptis, C.; Juraske, R.; Pfister, S.; Stoessel, F.; Antón, A.; Castells, F.; Hellweg, S. Biodiversity Impacts from Salinity Increase in a Coastal Wetland. *Environ. Sci. Technol.* **2013**, *47* (12), 6384–6392. https://doi.org/10.1021/es3045423.

(7) Perret, S. R.; Payen, S. Irrigation and the Environmental Tragedy: Pathways Towards Sustainability in Agricultural Water Use. *Irrig. Drain.* **2020**, *69* (2), 263–271. https://doi.org/10.1002/ird.2404.

(8) Elliott, M.; Day, J. W.; Ramachandran, R.; Wolanski, E.; Fang, Q.; Sheehan, M. R.; Seen, A. J.; Ellison, J. C. A Synthesis: What Is the Future for Coasts, Estuaries, Deltas and Other Transitional Habitats in 2050 and Beyond? In *Coasts and Estuaries: The Future*; Wolanski, E., Day, J. W., Elliott, M., Ramachandran, R., Eds.; Elsevier Inc., 2019; pp 1–28. https://doi.org/10.1016/B978-0-12-814003-1.00001-0.

(9) Zhou, J.; Chang, V. W. C.; Fane, A. G. An Improved Life Cycle Impact Assessment (LCIA) Approach for Assessing Aquatic Eco-Toxic Impact of Brine Disposal from Seawater Desalination Plants. *Desalination* **2013**, *308*, 233–241.

https://doi.org/10.1016/j.desal.2012.07.039.

(10) Böhme, D. Evaluation of Brine Discharge to Rivers and Streams: Methodology of Rapid Impact Assessment. *Limnologica* **2011**, *41*, 80–89. https://doi.org/10.1016/j.limno.2010.08.003.

(11) Parada, J. M.; Molares, J.; Otero, X. Multispecies Mortality Patterns of Commercial Bivalves in Relation to Estuarine Salinity Fluctuation. *Estuaries and Coasts* **2012**, 132–142. https://doi.org/10.1007/s12237-011-9426-2.

(12) Payen, S.; Basset-Mens, C.; Núñez, M.; Follain, S.; Grünberger, O.; Marlet, S.; Perret, S.; Roux, P. Salinisation Impacts in Life Cycle Assessment: A Review of Challenges and Options towards Their Consistent Integration. *Int. J. Life Cycle Assess.* **2016**, *21* (4), 577–594. https://doi.org/10.1007/s11367-016-1040-x.

(13) Leske, T.; Buckley, C. Towards the Development of a Salinity Impact Category for South African Life Cycle Assessments: Part 1 – A New Impact Category. *Water SA* **2003**, *30* (2), 253–265. https://doi.org/10.4314/wsa.v30i2.5070.

(14) Leske, T.; Buckley, C. Towards the Development of a Salinity Impact Category for South African Life Cycle Assessments: Part 2 - A Conceptual Multimedia Environmental Fate and Effect Model. *Water SA* **2004**, *30* (2), 241–251. https://doi.org/10.4314/wsa.v30i2.5070.

(15) Leske, T.; Buckley, C. Towards the Development of a Salinity Impact Category for South African Life Cycle Assessments: Part 3 - Salinity Potentials. *Water SA* **2004**, *30* (2), 253–265. https://doi.org/10.4314/wsa.v30i2.5071.

(16) Mosquera-Corral, A.; Campos Gómez, J. L. *Treatment and Valorisation of Saline Wastewater: Principles and Practice*; IWA Publishing, 2021. https://doi.org/https://doi.org/10.2166/9781789060645.

(17) Feitz, A. J.; Lundie, S. Soil Salinisation: A Local Life Cycle Assessment Impact Category. *Int. J. Life Cycle Assess.* **2002**, *7* (4), 244–249. https://doi.org/10.1007/BF02978880.

(18) Núñez, M.; Finkbeiner, M. A Regionalised Life Cycle Assessment Model to Globally Assess the Environmental Implications of Soil Salinization in Irrigated

Agriculture. *Environ. Sci. Technol.* **2020**, *54* (6), 3082–3090. https://doi.org/10.1021/acs.est.9b03334.

(19) Owsianiak, M.; Fantke, P.; Posthuma, L.; Saouter, E.; Vijver, M.; Backhaus, T.; Schlekat, T.; Hauschild, M. Chapter 7: Ecotoxicity. In *Global Guidance on Environmental Life Cycle Impact Assessment Indicators. Volume 2.*; Frischknecht, R., Jolliet, O., Eds.; UNEP, 2019; pp 138–172.

(20) Kingsford, R. T.; Brandis, K.; Bino, G.; Keith, D. A. *Freshwater Biome of the World*; Elsevier, 2020; Vol. 1. https://doi.org/10.1016/b978-0-12-409548-9.12479-0.

(21) Alvarez, I.; DeCastro, M.; Gomez-Gesteira, M.; Prego, R. Inter- and Intra-Annual Analysis of the Salinity and Temperature Evolution in the Galician Rías Baixas-Ocean Boundary (Northwest Spain). *J. Geophys. Res.* **2005**, *110* (4), 1–14. https://doi.org/10.1029/2004JC002504.

(22) Carral, L.; Barros, J. J. C.; Fidalgo, H. C.; Fabal, C. C.; Doce, A. M. Greenhouse Gas Emissions and Energy Consumption of Coastal Ecosystem Enhancement Programme through Sustainable Artificial Reefs in Galicia. *Int. J. Environ. Res. Public Health* **2021**, *18* (4), 1–19. https://doi.org/10.3390/ijerph18041909.

(23) Parada, J. M.; Molares, J. Natural Mortality of the Cockle Cerastoderma Edule (L.) from the Ria of Arousa (NW Spain) Intertidal Zone. **2008**, *43* (3), 501–511.

(24) Domínguez, R.; Vázquez, E.; Woodin, S. A.; Wethey, D. S.; Peteiro, L. G.; Macho, G.; Olabarria, C. Sublethal Responses of Four Commercially Important Bivalves to Low Salinity. *Ecol. Indic.* **2020**, *111* (January), 106031. https://doi.org/10.1016/j.ecolind.2019.106031.

(25) Peteiro, L. G.; Woodin, S. A.; Wethey, D. S.; Costas-costas, D.; Martínez-casal, A.; Olabarria, C.; Vázquez, E. Responses to Salinity Stress in Bivalves: Evidence of Ontogenetic Changes in Energetic Physiology on Cerastoderma Edule. *Sci. Rep.* **2018**, No. May, 1–9. https://doi.org/10.1038/s41598-018-26706-9.

(26) Roibás-Rozas, A.; Mosquera-Corral, A.; Hospido, A. Environmental Assessment of Complex Wastewater Valorisation by Polyhydroxyalkanoates Production. *Sci. Total Environ.* **2020**, *744*. https://doi.org/10.1016/j.scitotenv.2020.140893.

(27) Rosenbaum, R. K.; Hauschild, M. Z.; Boulay, A.-M.; Fantke, P.; Laurent, A.; Núñez, M.; Vieira, M. Life Cycle Impact Assessment. In *Life Cycle Assessment. Theory and Practice.*; Hauschild, M. Z., Rosenbaum, R. K., Olsen, S. I., Eds.; Springer, 2018. https://doi.org/10.1007/978-3-319-56475-3_10.

(28) Fantke, P.; Bijster, M.; Guignard, C.; Hauschild, M.; Huijbregts, M.; Jolliet, O.; A., K.; Magaud, V.; Margni, M.; TE., M.; Posthuma, L.; Rosenbaum, R. K.; van de Meent, D.; Zelm, V. *USEtox 2.0 Documentation (Version 1)*; Fantke, P., Ed.; 2018. https://doi.org/10.11581/DTU:00000011.

(29) Ziemann, H.; Schulz, C. Methods for Biological Assessment of Salt-Loaded Running Waters – Fundamentals, Current Positions and Perspectives. *Limnologica* **2011**, *41* (2), 90–95. https://doi.org/10.1016/j.limno.2010.09.005.

(30) Piscart, C.; Kefford, B. J.; Beisel, J. Are Salinity Tolerances of Non-Native Macroinvertebrates in France an Indicator of Potential for Their Translocation in a New Area%? *Limnologica* **2011**, *41* (2), 107–112. https://doi.org/10.1016/j.limno.2010.09.002.

(31) Kefford, B. J.; Marchant, R.; Schäfer, R. B.; Metzeling, L.; Dunlop, J. E.; Choy, S. C.; Goonan, P.; Victoria, E. P. A. The Definition of Species Richness Used by Species Sensitivity Distributions Approximates Observed Effects of Salinity on Stream Macroinvertebrates. *Environ. Pollut.* **2011**, *159* (1), 302–310. https://doi.org/10.1016/j.envpol.2010.08.025.

(32) Verbrugge, L. N. H.; Schipper, A. M.; Huijbregts, M. A. J.; Velde, G. Van der; Leuven, R. S. E. W. Sensitivity of Native and Non-Native Mollusc Species to Changing River Water Temperature and Salinity. *Biol Invasions* **2012**, 1187–1199. https://doi.org/10.1007/s10530-011-0148-y.

(33) Du, J.; Park, K.; Jensen, C.; Dellapenna, T. M.; Zhang, W. G.; Shi, Y. Massive Oyster Kill in Galveston Bay Caused by Prolonged Low-Salinity Exposure after Hurricane Harvey. *Sci. Total Environ.* **2021**, *774*, 145132. https://doi.org/10.1016/j.scitotenv.2021.145132.

(34) Johnson, A. J.; Shields, E. C.; Kendrick, G. A.; Orth, R. J. Recovery Dynamics of the Seagrass Zostera Marina Following Mass Mortalities from Two Extreme

Climatic Events. *Estuaries and Coasts* **2021**, *44* (2), 535–544. https://doi.org/10.1007/s12237-020-00816-y.

(35) Fox, D. R.; van Dam, R. A.; Fisher, R.; Batley, G. E.; Tillmanns, A. R.; Thorley, J.; Schwarz, C. J.; Spry, D. J.; McTavish, K. Recent Developments in Species Sensitivity Distribution Modeling. *Environ. Toxicol. Chem.* **2021**, *40* (2), 293–308. https://doi.org/10.1002/etc.4925.

(36) Smyth, K.; Elliot, M. Effects of Changing Salinity on the Ecology of the Marine Environment. In *Stressors in the Marine Environment: Physiological and ecological responses; societal implications*; Solan, M., Whiteley, N., Eds.; Oxford Scholarship, 2016; pp 161–174. https://doi.org/DOI:10.1093/acprof:oso/9780198718826.003.0009.

(37) Ministerio de Agricultura Pesca y Alimentación. GeoPortal https://sig.mapama.gob.es/geoportal/ (accessed Mar 29, 2021).

(38) Augas de Galicia. Rede de aforos https://augasdegalicia.xunta.gal/portada (accessed Mar 29, 2021).

(39) Ministerio para la transición ecológica y el reto demográfico. *Informe 2018 Autorizaciones de Vertido*; 2018.

(40) Dirección Xeral de Calidade e Avaliación Ambiental. *Resolución de 27 de Xuño de 2008, 2006/0347_NAA/IPPC*; 2008.

(41) Barros, M. C.; Magán, A.; Valiño, S.; Bello, P. M.; Casares, J. J.; Blanco, J. M. Identification of Best Available Techniques in the Seafood Industry: A Case Study. *J. Clean. Prod.* **2009**, *17* (3), 391–399. https://doi.org/10.1016/j.jclepro.2008.08.012.

(42) Bello Bugallo, P. M.; Stupak, A.; Cristóbal Andrade, L.; Torres López, R. Material Flow Analysis in a Cooked Mussel Processing Industry. *J. Food Eng.* **2012**, *113* (1), 100–117. https://doi.org/10.1016/j.jfoodeng.2012.05.014.

(43) Tomczak-Wandzel, R.; Vik, E. A.; Wandzel, T. *BAT in Fish Processing Industry*; 2015. https://doi.org/10.6027/tn2015-566.

(44) Mcjannet, D. L.; Webster, I. T.; Cook, F. J. Environmental Modelling & Software An Area-Dependent Wind Function for Estimating Open Water Evaporation Using Land-Based Meteorological Data. *Environ. Model. Softw.* **2012**, *31*, 76–83. https://doi.org/10.1016/j.envsoft.2011.11.017.

(45) Risch, E.; Loubet, P.; Núñez, M.; Roux, P. How Environmentally Significant Is Water Consumption during Wastewater Treatment?: Application of Recent Developments in LCA to WWT Technologies Used at 3 Contrasted Geographical Locations. *Water Res.* **2014**, *7*. https://doi.org/10.1016/j.watres.2014.03.023.

(46) MeteoGalicia. Rede Meteorolóxica https://www.meteogalicia.gal/observacion/estacions/estacions.action?request_locale=gl (accessed Mar 30, 2021).

(47) Fekete, B. M.; Vörösmarty, C. J.; Grabs, W. UNH/GRDC Composite Runoff Fields V1.0 https://www.compositerunoff.sr.unh.edu/index.html (accessed Mar 29, 2021).

(48) Venâncio, A.; Montero, P.; Costa, P.; Regueiro, S.; Brands, S.; Taboada, J. An Integrated Perspective of the Operational Forecasting System in Rías Baixas (Galicia, Spain) with Observational Data and End-Users. In *Computational Science - ICCS 2019*; Springer: Faro, Portugal, 2019. https://doi.org/10.1007/978-3-030-22747-0.

(49) Estévez, E.; Rodríguez-Castillo, T.; González-Ferreras, A. M.; Cañedo-Argüelles, M.; Barquín, J. Drivers of Spatio-Temporal Patterns of Salinity in Spanish Rivers: A Nationwide Assessment. *Philos. Trans. R. Soc. B Biol. Sci.* **2019**, *374* (1764), 1–10. https://doi.org/10.1098/rstb.2018.0022.

(50) Dias, R. da S.; Naves, A.; Pisani, B.; García-Tomillo, A. Comparación de La Concentración y Del Lixiviado de Cloruro y Sulfato En Dos Cuencas Con Diferente Distancia a La Costa. Conference Proceeding, Vol XIV. In *Jornadas de la zona no saturada*; Madrid, Spain, 2019; p 10.

(51) Alcalá, F. J.; Custodio, E. Using the Cl/Br Ratio as a Tracer to Identify the Origin of Salinity in Aquifers in Spain and Portugal. *J. Hydrol.* **2008**, *359* (1–2), 189–207. https://doi.org/10.1016/j.jhydrol.2008.06.028.

(52) WHO. *WHO Guidelines for Drinking-Water Quality*; 2006.

(53) Carregosa, V.; Velez, C.; Soares, A. M. V. M.; Figueira, E.; Freitas, R. Physiological and Biochemical Responses of Three Veneridae Clams Exposed to Salinity Changes. *Comp. Biochem. Physiol. Part B* **2014**, *177–178*, 1–9. https://doi.org/10.1016/j.cbpb.2014.08.001.

(54) Gharbi, A.; Farcy, E.; Van Wormhoudt, A.; Denis, F. Response of the Carpet Shell Clam (Ruditapes Decussatus) and the Manila Clam (Ruditapes Philippinarum) to Salinity Stress. *Biologia (Bratisl).* **2016**, *71*, 551–562. https://doi.org/10.1515/biolog-2016-0072.

(55) Verdelhos, T.; Marques, J. C.; Anastácio, P. The Impact of Estuarine Salinity Changes on the Bivalves Scrobicularia Plana and Cerastoderma Edule , Illustrated by Behavioral and Mortality Responses on a Laboratory Assay. *Ecol. Indic.* **2015**, *52*, 96–104. https://doi.org/10.1016/j.ecolind.2014.11.022.

(56) Fiett, P. P. Salinity Tolerance and Osmoregulatory Function of Mannitol in Danish Ecotype of Saccharina Latissima. 1–24.

(57) Peteiro, C.; Sánchez, N. Comparing Salinity Tolerance in Early Stages of the Sporophytes of a Non Indigenous Kelp (Undaria Pinnatifida) and a Native Kelp (Saccharina Latissima). *J. Mar. Biol.* **2012**, *38* (2), 197–200. https://doi.org/10.1134/S1063074012020095.

(58) Freitas, R.; Pires, A.; Velez, C.; Almeida, Â.; Wrona, F. J.; Soares, A. M. V. M.; Figueira, E. The Effects of Salinity Changes on the Polychaete Diopatra Neapolitana: Impacts on Regenerative Capacity and Biochemical Markers. *Aquat. Toxicol.* **2015**, *163*, 167–176. https://doi.org/10.1016/j.aquatox.2015.04.006.

(59) Reyes-martínez, M. J.; Martínez-pita, I.; Soler-navarro, D.; García-garcía, F. J. The Impact of Salinity Changes Associated with Size on the Wedge Clam Donax Trunculus Linnaeus, 1758 (Mollusca Bivalvia): A Laboratory Assay. *Estuar. , Coast. Shelf Sci.* **2020**, *241*, 106838. https://doi.org/https://doi.org/10.1016/j.ecss.2020.106838.

(60) Gaag, M. Van Der; Velde, G. Van Der; Wijnhoven, S. Salinity as a Barrier for

Ship Hull‐Related Dispersal and Invasiveness of Dreissenid and Mytilid Bivalves. *Mar. Biol.* **2016**, *163* (7), 1–13. https://doi.org/10.1007/s00227-016-2926-7.

(61) Norton, T. A.; South, G. R. Influence of Reduced Salinity on the Distribution of Two Laminarian Algae. *Nord. Soc. Oikos* **1969**, *20* (2), 320–326.

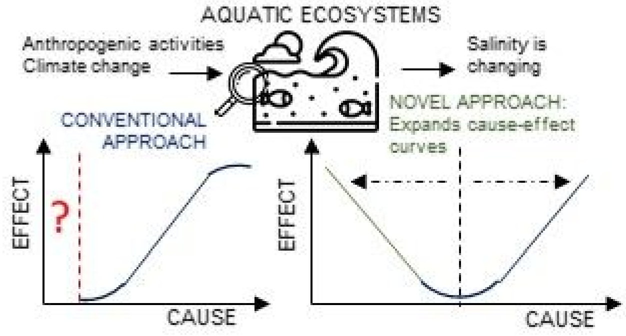
(62) Salo, T.; Foldager, M.; Boström, C. Population Specific Salinity Tolerance in Eelgrass (Zostera Marina). *J. Exp. Mar. Bio. Ecol.* **2014**, *461*, 425–429. https://doi.org/10.1016/j.jembe.2014.09.010.

(63) Fernández-Torquemada, Y.; Sánchez-Lizaso, J. L. Responses of Two Mediterranean Seagrasses to Experimental Changes in Salinity. *Hydrobiologia* **2011**, *699*, 21–33. https://doi.org/10.1007/s10750-011-0644-1.

(64) Berger, E.; Frör, O.; Schäfer, R. B. Salinity Impacts on River Ecosystem Processes: A Critical Mini-Review. *Philos. Trans. R. Soc. B Biol. Sci.* **2019**, *374* (1764). https://doi.org/10.1098/rstb.2018.0010.

(65) Verones, F.; Bare, J.; Bulle, C.; Frischknecht, R.; Hauschild, M.; Hellweg, S.; Henderson, A.; Jolliet, O.; Laurent, A.; Liao, X.; Lindner, J. P.; Maia de Souza, D.; Michelsen, O.; Patouillard, L.; Pfister, S.; Posthuma, L.; Prado, V.; Ridoutt, B.; Rosenbaum, R. K.; Sala, S.; Ugaya, C.; Vieira, M.; Fantke, P. LCIA Framework and Cross-Cutting Issues Guidance within the UNEP-SETAC Life Cycle Initiative. *J. Clean. Prod.* **2017**, *161*, 957–967. https://doi.org/10.1016/j.jclepro.2017.05.206.

For Table of Contents Only

784 
AQUATIC ECOSYSTEMS
Anthropogenic activities
Climate change
Salinity is changing
CONVENTIONAL APPROACH
NOVEL APPROACH: Expands cause-effect curves
EFFECT
?
CAUSE
EFFECT
CAUSE

785