The effects of mining pollution on subtidal habitats of northern Chile

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Abstract: Mining is the most important commercial activity in Chile, producing more than 60% of the total national income. In northern Chile, where mining extraction is concentrated, solid and liquid pollutants resulting from copper and iron ore processing are discharged to the ocean. The studies on the effects of mining pollution on coastal marine environments in Chile mainly consist of the evaluation of heavy metal concentrations in organisms and intertidal sediments. There are no references on the effects of mining discharges on populations and communities of the shallow subtidal habitats. During 1996–1997, covering more than 1500 km of coast (in polluted and control areas), we studied (1) the chemical characterization of copper and iron discharges, (2) the distribution of heavy metals in *Lessonia trabeculata* (frond, stipe and holdfast), (3) the distribution of heavy metals in seawater, (4) changes on *Lessonia* composition and quality of phycocolloid and morphology, (5) the effects upon the macroinvertebrate community associated with *Lessonia* holdfast and on the rocky subtidal community. All these variables were examined over time, at different depths and at various distances from the discharge source. We show here that tailings from copper mining cause more ecological perturbation than those from iron extraction. However, the lack of organisms in areas close to mining activities could be caused by the accumulation of fine sediments more than the heavy metal content *per se*. This work shows that the levels of heavy metals in seawater, plants and alginates of *Lessonia* in contaminated and pristine sites are highly variable. In this context, environmental factors, such as temperature, wind intensity, tidal regimes, water movement, wave impact, coastal circulation, local orogenetic processes, tectonic movement, shore topography, coastal upwelling and global oceanographic phenomena, such as El Niño, must be considered in the future in the interpretation of the effects of heavy metals in coastal marine communities.

Keywords:

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1 Introduction

Chile is a country of enormous mining potential, in which copper is the main export, accounting for over 60% of the total income. Between 18° and 30° S, there are enormous deposits, including over 25% of the copper, 40% of the molybdenum, and 30% of the lithium world reserves, plus important deposits of Au, Fe, Ag, Mn, Co, Hg, Tu, Pb and Zn (Corvalán, 1985). The extraction of these non-renewable resources leads to severe modifications of terrestrial ecosystems, not only by the removal of material but also by the use of superficial and underground waters, the occupation of lands and the discharge of solid and liquid residues. These residues, with a high content in heavy mètals, are deposited in dams or directly dumped into water courses or into the sea, producing catastrophic perturbations in terrestrial and marine biota.

In the studies of the effects of mining tailings on littoral marine communities in the South-East Pacific, the trend is to refer almost exclusively to the evaluation of heavy metals in different organisms (algae, invertebrates and fish) and sediments (Boré *et al.*, 1989; Trucco *et al.*, 1990; Vermeer and Castilla, 1991; Lecaros and Astorga, 1992; Ahumada, 1994; Vásquez and Guerra, 1996). Most of these studies are difficult to contrast owing to the methodologies employed and the lack of data on non-contaminated areas. This is particularly relevant in this region, where orogenetic processes, volcanic activity and climatic conditions naturally increase the availability of heavy metals (Vila and Sillitoe, 1991; Vásquez and Guerra, 1996).

Although there is evidence of natural catastrophes produced by the accumulation of fine sediments resulting from Cu and Fe mining (e.g. Chañaral in northern Chile, 25 000 ton/day from 1939 to 1975), only a few drastic effects of the tailings have been reported for the marine coastal communities of northern Chile. Apart from the studies of Castilla and Nealler (1978), Castilla (1983) and Vásquez and Guerra (1996), which reported changes in the species composition of intertidal communities affected by Cu tailings, no data are to be found in the literature on the effect of such discharges in subtidal communities.

The lack of data on the ecological effects of heavy metals, aside from those reported in extreme cases (Bryan and Langston, 1992), can result from the way in which such effects have been studied. Although ecotoxicological studies in the laboratory can identify the effects of contaminants under controlled conditions, they cannot reproduce the incidence of the number of environmental factors on the magnitude of these effects. Therefore, what occurs in the field can differ widely from the potential effects identified under laboratory conditions. In this context, through manipulative experiments in subtidal soft bottoms, Morrisey *et al.* (1996) increased the amount of Cu ions in the sediments, demonstrating that this cation affects the species composition in this coastal marine community. The nature of the response depends on the taxa analysed, on the relative abundance of the species present, on the time of the observations, and on the local environmental conditions. In this context, the temporal variations of the in-fauna (Cu treatment vs control) were not always consistent between the replicates of the experimental areas, and between different spatial scales (from 5 to 100 m).

Between 18° and 42° S, the shallow subtidal environments of rocky bottoms (0–35 m in depth) are dominated by *Lessonia trabeculata* (Phaeophyta, Laminariales) (Villouta and Santelices, 1986; Vásquez, 1992). These environments, exposed to water movement, have been the most affected by tailings of Cu and Fe mining in northern Chile. Considering the logistic problems met when experimentally 'injecting' heavy metals in exposed rocky environments (Morrisey *et al.*, 1986), a valid approach to study experimentally the effect of mining tailings in these communities is to determine the distribution of the contaminants in a gradient from the contamination focus (assuming a dilution effect), correlatively evaluating their effects on populations and/or communities. This natural experiment should allow the determination of the *in situ* effect of mining tailings, without isolating co-variation effects from natural phenomena such as temperature, salinity, pH, wave exposure, and water movement. Further, it incorporates a gradient of the magnitude of the perturbations to which the biological communities are subjected in a natural environment.

The present work characterizes the subtidal environments contaminated by Cu and Fe mining through the quantification of these cations in seawater and in *L. trabeculata* (plants and alginates). It also evaluates the temporal and spatial effects of Cu and Fe tailings on rocky subtidal populations and communities dominated by macroalgae in northern Chile, in distance and depth gradients established from the source of the perturbation.

2 Materials and methods

Rocky subtidal environments affected by solid and liquid tailings from Cu (Michilla, 22° 48' S) and Fe mining (Chapaco, 28° 28' S), were seasonally sampled between June 1996 and August 1997. Simultaneously, two localities 60 km north of Chapaco (Carrizal Bajo, 28° 05' S) and 100 km south of Michilla (Caleta Constitución, 23° 25' S) were evaluated as control areas (Figure 1). Samples of seawater and of the dominant subtidal populations and communities were taken in the study areas, at permanent stations located on a distance gradient from the contamination source (0, 1, 2, 3, and 5 km) and on a depth gradient (0, 10, 20, and 30 m). The subtidal localities correspond to rocky areas exposed to predominant southwesterly winds, with northerly coastal currents and with bottom communities dominated by *L. trabeculata* (Vásquez, 1992).

Chemically, the study areas were characterized by the evaluation in seawater of Cu^{2+} (Michilla) and Fe^{2+} (Chapaco) from samples obtained by scuba diving on the bottom (in the depth gradient), and at 10 m depth for each sampling station in the distance gradient. The dams and ducts that evacuate mining residues to the shore were simultaneously evaluated in all study areas.

2.1 *Heavy metals in seawater*

The preconcentration of the seawater samples was carried out according to Berndt *et al.* (1985). A 500 ml sample was filtered with a Millipore 0.45 μm filter at pH 4–5 (sodium acetate–acetic acid) and treated with 80 mg of 1-ammonium pyrrolidine-dithiocarbamate (APDTC) dissolved in 2 ml of water. It was then vigorously stirred and filtered drop by drop through active carbon. The metallic complexes absorbed by the active carbon were dried at 120 °C for 20 min. After the system had cooled, it was treated with 1 ml of HNO_3, heated and dried. Finally, the carbonaceous residue with the metallic cations was suspended in 1.5 ml of HNO_3 (4.5 M) and centrifuged at 10 000 rpm for 15 min. Analytical readings were done with 200 μl aliquots of each sample by atomic absorption spectroscopy, in a 2380 Perkin Elmer, using as standards 1000 μg/ml metallic Cu and Fe in 0.3 M HNO_3 (J.T. Baker, INSTRA-ANALYZED Reagent).

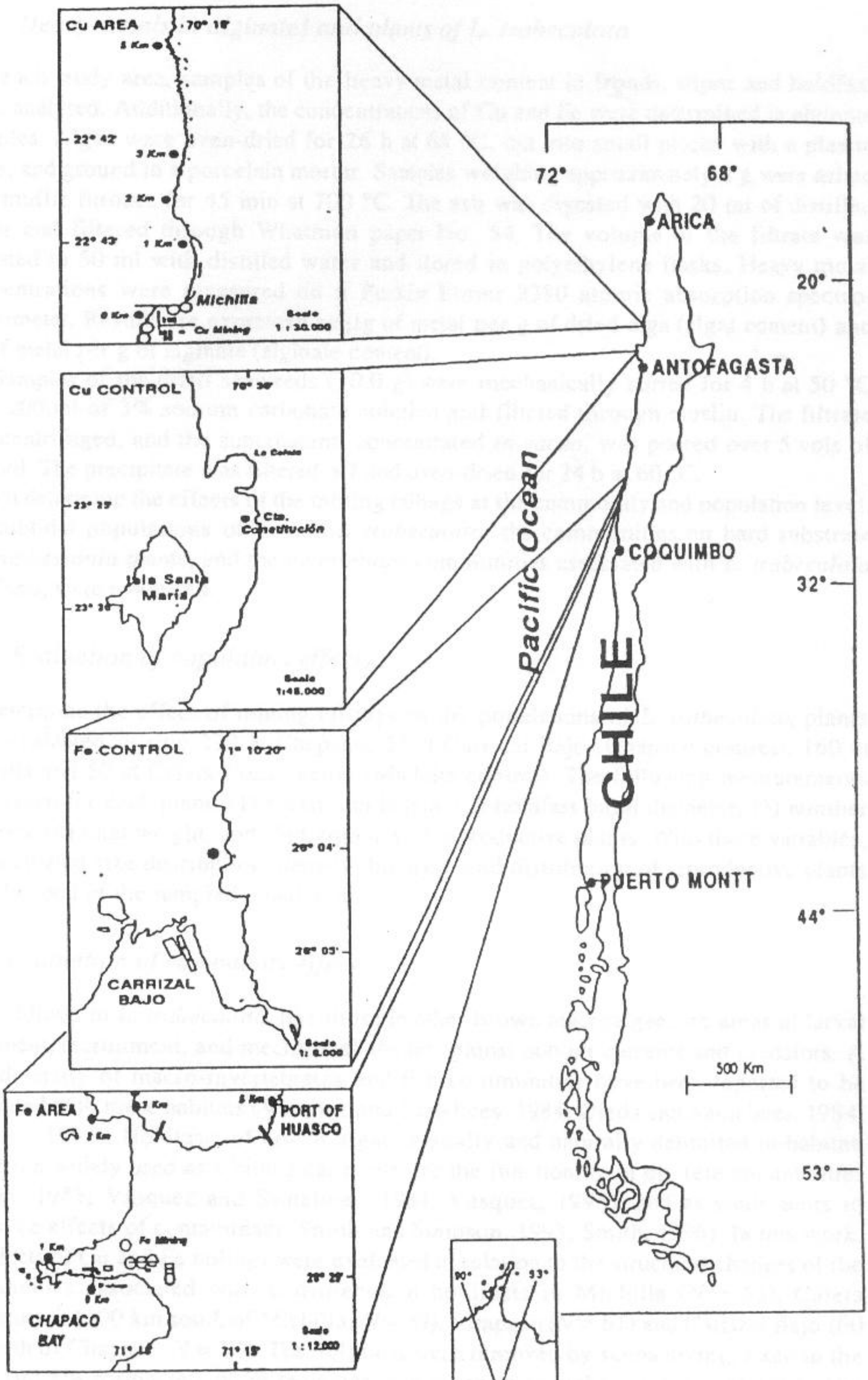


Figure 1 The two sampling areas and the two control areas.

2.2 *Heavy metals in alginates and plants of L. trabeculata*

For each study area, samples of the heavy metal content in fronds, stipes and holdfast were analysed. Additionally, the concentrations of Cu and Fe were determined in alginate samples. Algae were oven-dried for 26 h at 65 °C, cut into small pieces with a plastic knife, and ground in a porcelain mortar. Samples weighing approximately 2 g were ashed in a muffle furnace for 45 min at 700 °C. The ash was digested with 20 ml of distilled water and filtered through Whatman paper No. 54. The volume of the filtrate was adjusted to 50 ml with distilled water and stored in polyethylene flasks. Heavy metal concentrations were measured on a Perkin Elmer 2380 atomic absorption spectrophotometer. Results are expressed as µg of metal per g of dried alga (algal content) and µg of metal per g of alginate (alginate content).

Samples of the dried seaweeds (10.0 g) were mechanically stirred for 4 h at 50 °C with 200 ml of 3% sodium carbonate solution and filtered through muslin. The filtrate was centrifuged, and the supernatant, concentrated *in vacuo*, was poured over 5 vols of ethanol. The precipitate was filtered off and oven-dried for 24 h at 60 °C.

To determine the effects of the mining tailings at the community and population level, the subtidal populations of *Lessonia trabeculata*, the communities on hard substrate among *Lessonia* plants, and the invertebrate communities associated with *L. trabeculata* holdfasts, were evaluated.

2.3 *Evaluation of population effects*

To determine the effect of mining tailings on the populations of *L. trabeculata*, plants were evaluated *in situ*: 339 at Chapaco, 57 at Carrizal Bajo (Chapaco-control), 160 at Michilla and 50 at Caleta Constitución (Michilla control). The following measurements were taken for each plant: (1) maximum length, (2) holdfast basal diameter, (3) number of stipes, (4) total weight, and (5) frequency of reproductive plants. With these variables, we calculated size distribution, density, biomass, and distribution of reproductive plants over the total of the sampled population.

2.4 *Evaluation of community effects*

The holdfasts in *L. trabeculata*, like those in other brown macroalgae, are areas of larval settlement, recruitment, and mechanical shelter against bottom currents and predators. A high diversity of macro-invertebrates and fish communities have been reported to be associated with these habitats (Vásquez and Santelices, 1984; Ojeda and Santelices, 1984; Vásquez, 1993). Holdfasts of brown algae, spatially and naturally delimited in habitat, have been widely used as a biological model for the functioning of discrete communities (Snider, 1985; Vásquez and Santelices, 1984; Vásquez, 1993), and as study units to assess the effects of contaminant (Smith and Simpson, 1993; Smith, 1996). In this work, the effects of Cu and Fe tailings were evaluated in relation to the structural changes of the communities associated with *L. trabeculata* holdfasts in Michilla (N = 53), Caleta Constitución (100 km south of Michilla, N = 49), Chapaco (N = 61) and Carrizal Bajo (60 km north of Chapaco, N = 57). The holdfasts were removed by scuba diving, fixed in the field with 8% formaldehyde in seawater, and taken to the laboratory for analysis. The fauna were extracted from the central cavity, identified when possible to the species level, counted, weighed and measured.

In each study area, and throughout the distance gradient from the contamination source, the communities from hard bottoms among *L. trabeculata* plants were evaluated using transects of 50 m long parallel to the shore, at 10 m depth. The transects (Michilla $N = 6$, Chapaco $N = 8$) were sub-divided each 10 m, and the cover and/or density (two replicates at random every 10 m) was measured with 0.25 m^2 quadrats, with 100 intersection points.

The temporal and spatial changes of the subtidal communities were evaluated according to the changes in species richness, diversity, total density, total biomass and cover (communities between *Lessonia* plants). To estimate the differences in the population and community parameters, one-way ANOVA and *a posteriori* Tukey tests (Sokal and Rolf, 1981) were performed. The confidence intervals for diversity index were obtained by Jacknife (Jaksic and Medel, 1987).

To compare the effects of Cu and Fe tailings on the spatial and temporal gradients and to discriminate the variability inherent to population and community, the values of the abundance of the species present in *Lessonia trabeculata* holdfasts were transformed into root 4 and arranged by a non-parametric multivariate analysis, based on Bray–Curtis similarity coefficients (Ludwig and Reynolds, 1988). Differences between the sampling times and between localities were analysed by the ANOSIM test (Warwick *et al.*, 1988).

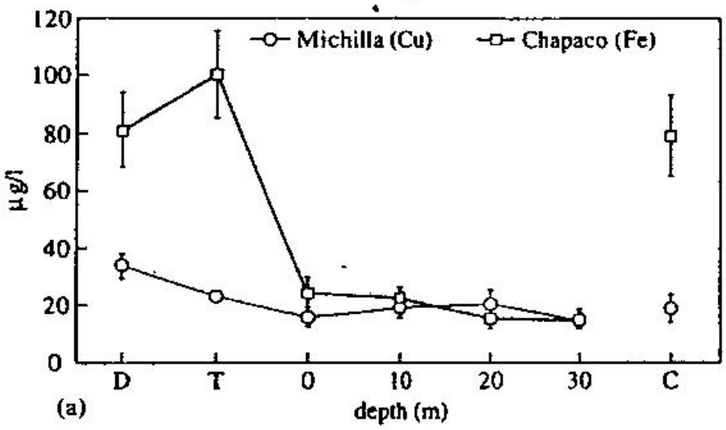
3 Results

3.1 *Heavy metals in seawater*

The spatial distribution of Cu^{2+} and Fe^{2+} in seawater in sites affected by mining tailings showed values that decreased with depth (0–30 km) and with distance (0–5 km) from the contamination focus (Figure 2). At Michilla, the values of Cu^{2+} in seawater reached their maximum mean values in the dam (D = 36 µg/l) and in the outfall (T = 25 µg/l) that evacuates tailings into the intertidal areas, where the Cu^{2+} content decreased to 18 µg/l. These values are maintained without significant variations ($P = 0.05$) until 30 m in depth (Figure 2a). Fe^{2+} in seawater showed a similar pattern at Chapaco, where maximum values occur in the dam and the outfall (80 and 100 µg/l, respectively). These values are significantly different ($P < 0.05$) from those observed between 0 and 30 m in depth, which fluctuate between 20 and 30 µg/l (Figure 2a).

Along a distance gradient between 0 and 5 km, the spatial variation of the concentrations Cu^{2+} and Fe^{2+} in seawater showed a clear dilution pattern in the first kilometre, at both localities studied (Figure 2b). Both Cu^{2+} and Fe^{2+} values had their maximum mean values at 0 m deep (intertidal areas), and significantly decreased ($P <$ 0.05) towards 1 km from the contamination source. However, between 1 and 5 km from the contamination source, the values did not significantly differ ($P = 0.05$) (Figure 2b).

Cu^{2+} and Fe^{2+} contents at control areas in Caleta Constitución (100 km from Michilla) and Carrizal Bajo (60 km from Chapaco) showed similar values to those of Cu^{2+} at 30 m deep, and significantly higher values ($P < 0.05$) than those of this cation 5 km from the contamination source. In the case of Fe^{2+}, concentrations observed in control areas (depth and distance), were significantly higher ($P < 0.05$) than those observed at the limits of the depth and distance gradients.



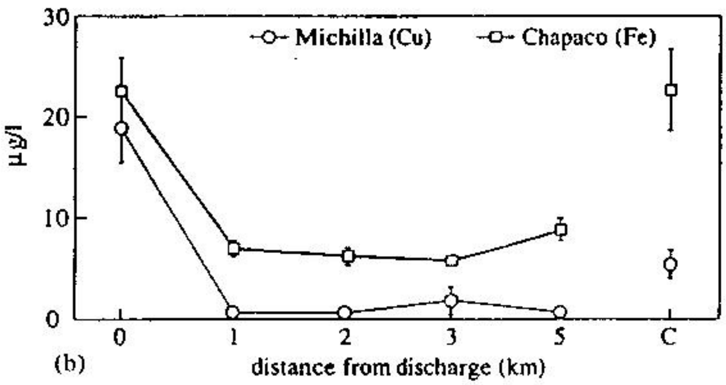


Figure 2 Variability of Cu^{2+} and Fe^{2+} contents in seawater in a depth gradient (a) and a distance gradient from the mining tailings (b).

Between 0 and 30 m depth, and between 0 and 5 km from the contamination source, the temporal distribution of Cu^{2+} and Fe^{2+} in seawater did not show a clear pattern during the study period in the areas of Michilla and Chapaco (Figure 3). Comparatively, concentrations of these metals in seawater measured during the study period at each level showed more variability in depth than in distance from the mining tails. Both in Michilla and in Chapaco, Cu^{2+} and Fe^{2+} values increased toward the winter months (0 m depth, 0 km), hence the temporal variability of Cu^{2+} and Fe^{2+}, at the depths and distances studied, showed a relation with this seasonal increase in the cations. This trend in the temporal variability could have been related more to the resuspension of the bottom sediments (rich in heavy metals) resulting from the winter season, than to the higher contributions of heavy metals (from the mining) over that season of the year (Figure 3).

3.2 Heavy metals in plants and alginates

Concentrations of Cu^{2+} and Fe^{2+} in frond, stipes and holdfasts of *Lessonia trabeculata* and in alginates showed no clear pattern as a function of the cation content in seawater (distance from the contaminating source) (Figure 4). *Lessonia* showed variable maximum Cu^{2+} concentrations in the different morphological structures analysed. In Michilla, Cu^{2+}

values at 3 and 5 km from the contaminating source increased in the order frond > holdfast > stipe; at the control locality (100 km away) the order was holdfast > stipe > frond. Cu^{2+} concentrations in alginates extracted from different *Lessonia* morphological structures were comparatively less variable than Cu^{2+} concentrations directly measured in the same plant morphological structures. At Michilla, 3 km away from the contaminating source, the maximum mean concentrations of Cu^{2+} (20 µg/g) in alginates were observed in *Lessonia* fronds. In contrast, at 5 km from the focus, the highest mean values of Cu^{2+} in alginates were found in the stipes. At the control area, alginates extracted from holdfasts and fronds showed Cu^{2+} concentrations that did not differ significantly ($P > 0.05$), corresponding to the higher concentrations observed in the study area (20 µg/g), which differ significantly ($P < 0.05$) from the Cu^{2+} concentrations found in alginates extracted from the stipes in the same plants (Figure 4).

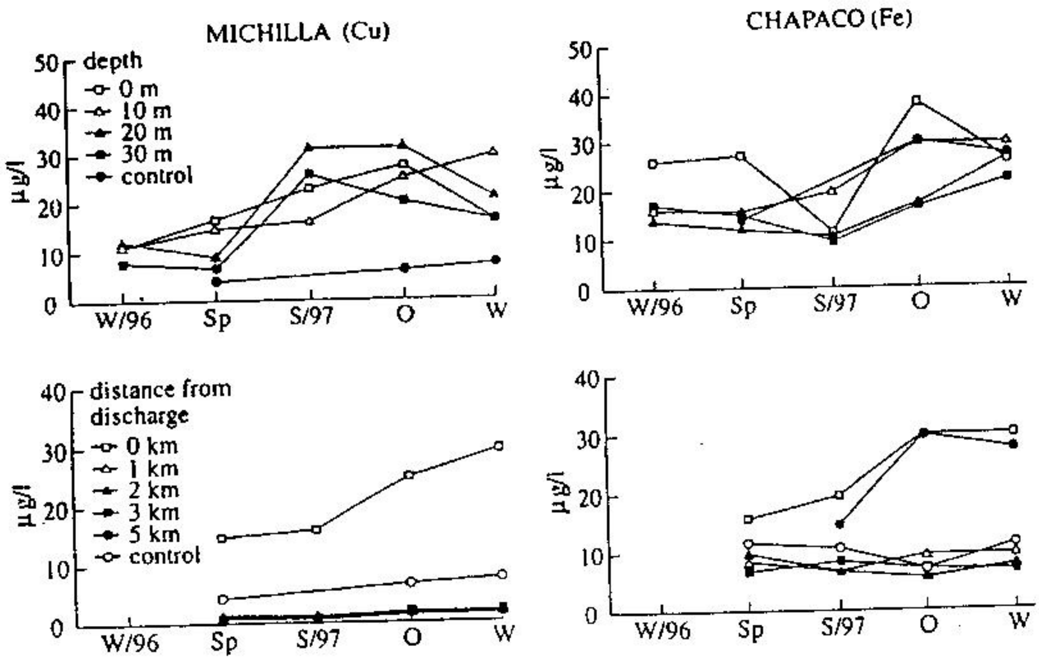


Figure 3 Temporal variation of Cu^{2+} and Fe^{2+} contents in seawater with depth and distance from the mining outfall. (W = winter; Sp = spring; S = summer).

At Chapaco, Fe^{2+} concentrations in the different *Lessonia* structures did not significantly vary with the distance from the discharge, except at 0 m and at the control site, where the highest concentrations were observed in the fronds. Note the high spatial and temporal variability of heavy metal concentrations (both in *Lessonia* and alginates) found in the study areas and controls (Figure 3). The highest concentrations of Fe^{2+} in alginates occurred in plants close to the tailings (0–1 km). Though no significant differences were found between Fe^{2+} concentrations in alginates at 2–5 km and between locality and control, a slight decrease of Fe^{2+} concentrations in alginates with the distance to the contamination source is observed. Consistently, the greatest Fe^{2+} concentrations occur in alginates from fronds, except at the 0 km where the highest Fe^{2+} values were found in extracts from the *L. trabeculata* holdfasts (Figure 4).

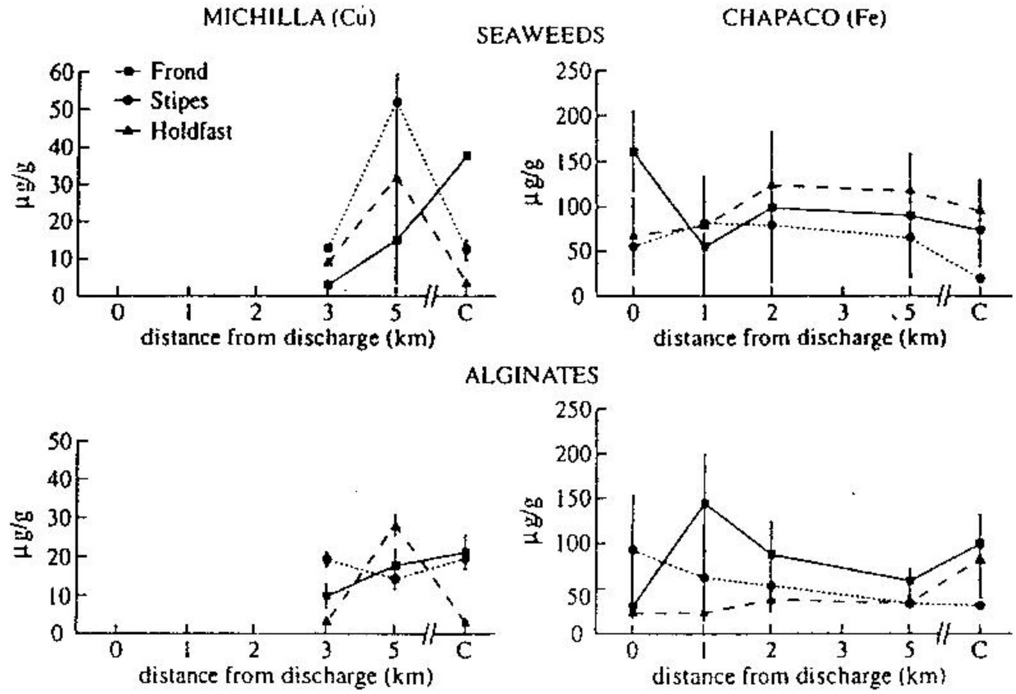


Figure 4 Contents of Cu^{2+} and Fe^{2+} in *L. trabeculata* (fronds, holdfast and stipe) plants and alginates in a distance gradient.

3.3 Population effects

At Michilla (Cu effect) and Chapaco (Fe effect), no *Lessonia trabeculata* plants were detected between 0 and 30 m depth. At this depth range (approx. 1 km), the rocky substrate is covered by a fine sediment, which precludes settlement and growth of spores and larvae from algae and invertebrates.

Considering the distance gradient at the localities studied, the first *Lessonia* plants appeared only at 3 km in Michilla, whereas a number of these plants were registered few metres from the outfall of Fe tailings (Figure 5). In Michilla, the population descriptors of *Lessonia* suggest that populations consist of small plants that do not exceed 1.8 m in length, 20 cm in holdfast diameter, 10 stipes per plant and 5 kg in mean wet weight. However plant density is high, over 60 plants/10 m^2 (Figure 5).

At Chapaco (Fe^{2+} effect), in the subtidal populations of *Lessonia trabeculata* in the distance gradient, plants exceed 2 m in length, 18 kg wet weight and 30 cm mean holdfast diameter (Figure 5). The mean density and the mean number of stipes per plant decrease with the distance to the contamination focus, suggesting an increase in plant size and a longer survival of adult plants owing to the decrease of the perturbation effect (Figure 5).

Comparatively, Cu^{2+} tailings produce greater perturbations at the population level than those of Fe^{2+}. The absence of *Lessonia* populations between 0 and 3 km at the locality of Michilla, and the size frequency as a function of the distance to the outfall showed a greater relative abundance of juvenile plants at 3 km, compared with a greater frequency of adult plants at 5 km (Figure 6). Consequently, the *Lessonia* populations farther from the contamination focus had a higher frequency of reproductive plants than those closer to it (Figure 6).

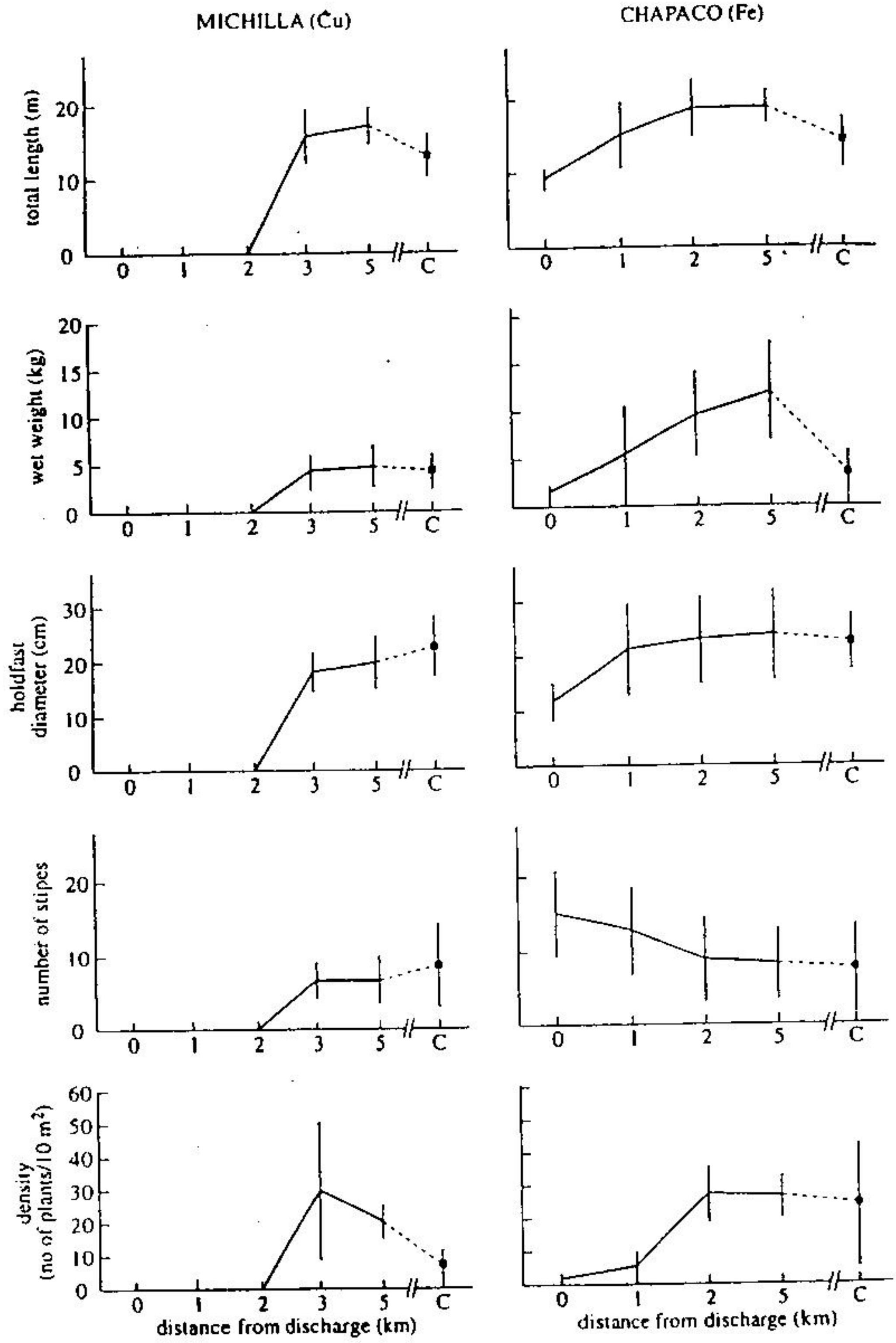


Figure 5 Spatial variation of variables of *L. trabeculata* at different exposures to Cu^{2+} and Fe^{2+}.

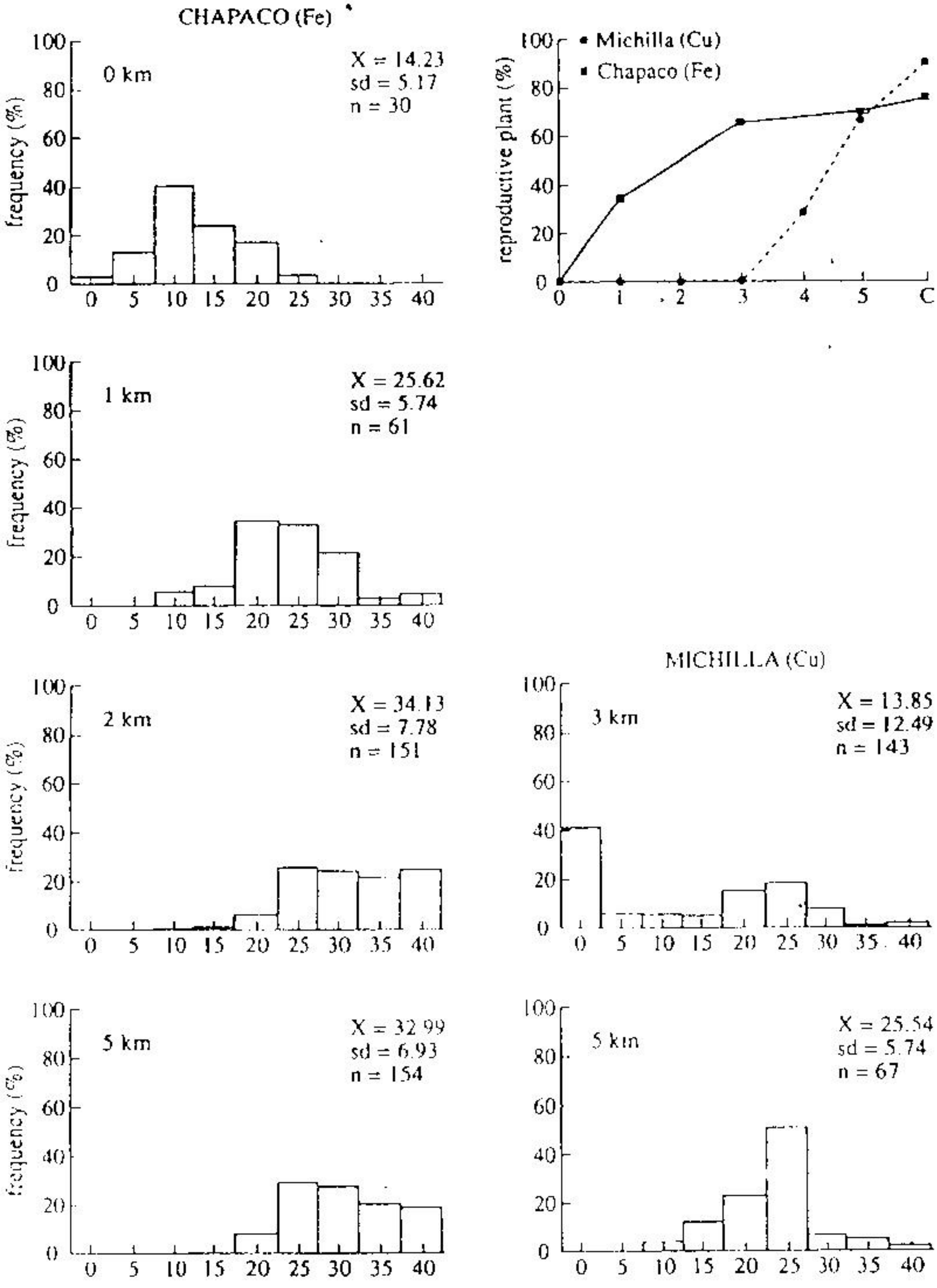


Figure 6 Spatial variation of size frequency and proportion of reproductive plants in subtidal *L. trabeculata* populations at different exposures to Cu^{2+} and Fe^{2+}.

3.4 Community effects

The communities of macroinvertebrates associated with *L. trabeculata* show a strong and significant ($P < 0.05$) reduction in species richness, density and biomass of the intra-holdfast organisms in areas closer to the contamination focus than in control areas. Comparatively, macro-invertebrate communities associated with *Lessonia* in environments contaminated by Fe tailings, show no significant differences ($P > 0.05$) from those in control areas, or from those inhabiting *Lessonia* holdfasts between 1 and 5 km from the contamination source (Figure 7). At the end of the distance gradient, in each study site and each control area, more than 130 species have been found inhabiting *Lessonia* holdfasts. Mollusca, Polychaeta and Arthropoda are the most diverse taxa in these microhabitats.

Communities on hard bottoms, between *Lessonia* plants, showed similar responses to those observed in intra-holdfast communities in both study sites, equivalent ecological effects and similar variability patterns in relation to the heavy metal levels in the environment (Figure 8). In this context, the greatest effects of mining pollution on hard bottom communities were observed in areas close to Cu and Fe discharges.

The results of multidimensional scaling (MDS), considering all samples during the study period (spatial and temporal variability of fauna inhabiting *Lessonia* holdfasts and inter-plant communities), do not show any grouping pattern at any of the localities (ANOSIM test, $P = 0.0999$), hence no community pattern may be suggested for each study area as a result of the Fe and Cu tailings. The STRESS values (Michilla = 0.2593 Chapaco = 0.4131) support this trend and suggest that this arrangement does not differ from the random one (Figure 9). In this context, excluding the effect in areas directly affected by the mining outfall, probably owing to sediments more than to heavy metals, the temporal and spatial variability observed in the localities studied are not different from the natural variability of the subtidal community analysed.

4 Discussion

Although it is found in the literature that contamination by heavy metals may cause dramatic ecological impacts on coastal marine environments, these effects are rarely documented except in cases of drastic contamination (Bryan and Langston, 1992). As suggested by Morrisey *et al.* (1996), the lack of evidence is partly due to the way in which these perturbations have been investigated. Traditionally, the effects of contaminants in marine environments have been approached in two ways. (1) Ecotoxicological studies that identify the effects of contaminants under controlled laboratory conditions; unfortunately such studies do not reproduce the range of potential environmental factors that incide on the magnitude of the effect, therefore the actual impact of these contaminants in nature widely differs from that reported under laboratory conditions. (2) Field studies, based on correlations between the distribution of the contaminants and the composition of the benthic fauna assemblages, which incorporate environmental variability. Generally, these studies do not include the intermediate effects proper to an environmental gradient of perturbation distribution, and the eventual 'responses of the population and/or communities' to this gradient. In this context, Bryan and Langston (1992) suggest that the effects of Cu and Zn on the species distribution, although evident, are not so dramatic and obvious as predicted from controlled laboratory conditions. For example, in the field, deleterious effects on benthic organisms that could be directly attributed to specific effects of metallic contaminants are very scarce compared with those reported for equivalent laboratory assays.

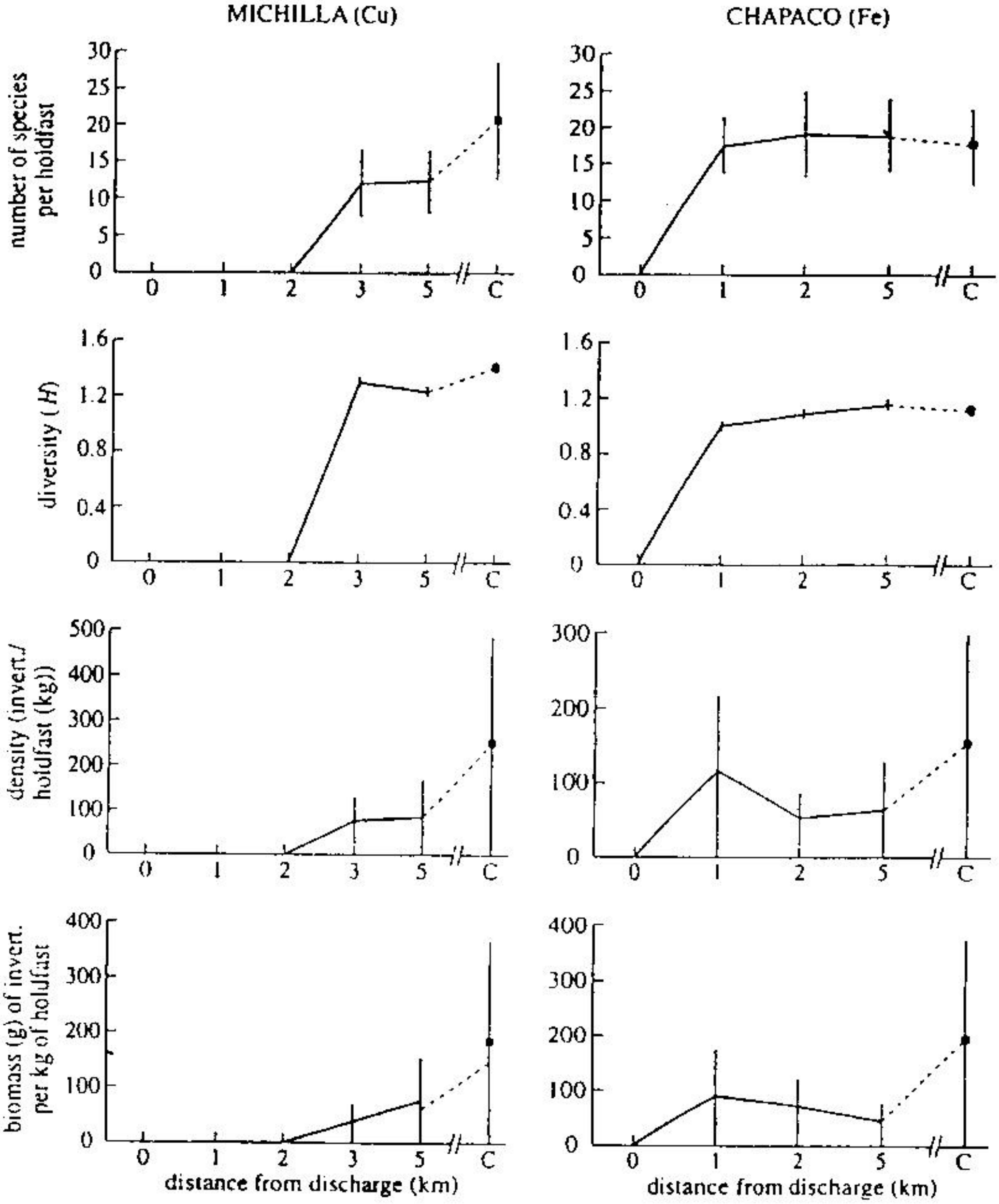


Figure 7 Spatial variation of macroinvertebrate fauna associated with *L. trabeculata* holdfasts at different exposure to Cu^{2+} and Fe^{2+}.

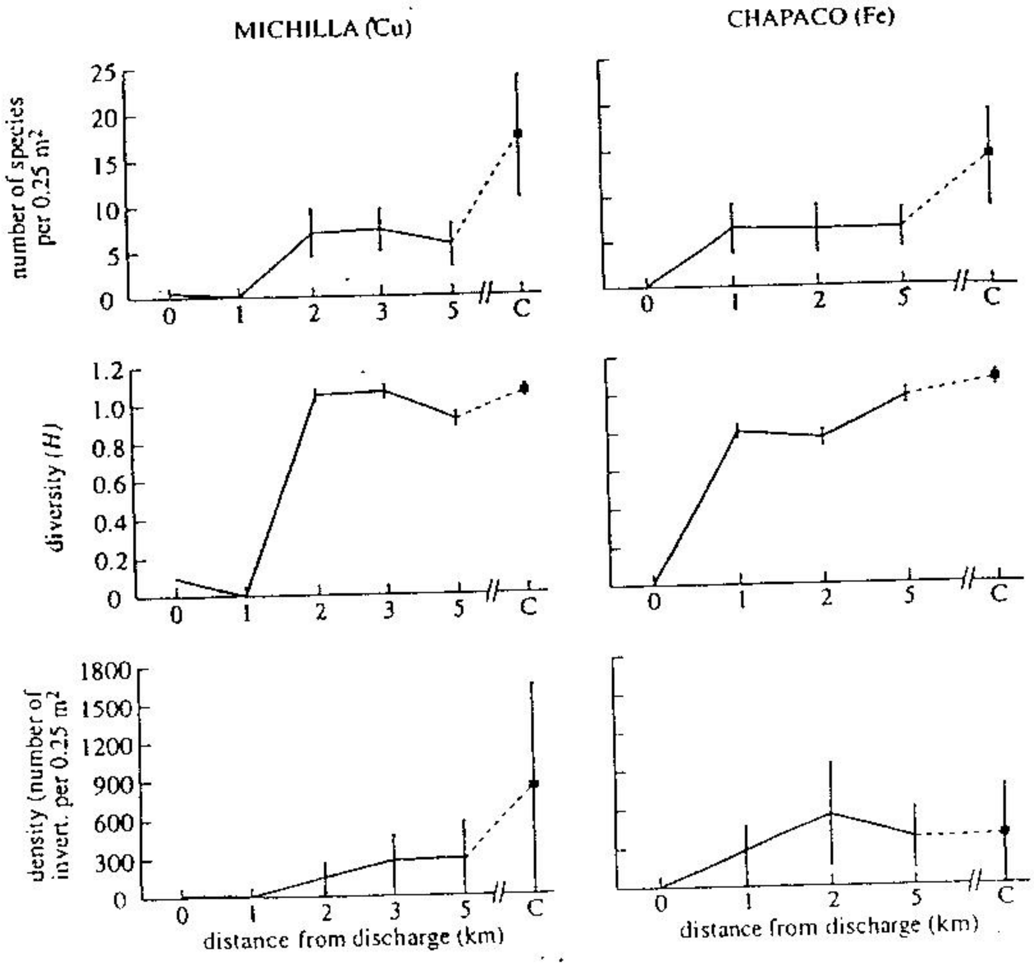


Figure 8 Spatial variation of hard bottom communities between *Lessonia* plants at different exposure to Cu^{2+} and Fe^{2+}.

Copper, an essential micro-nutrient for marine micro- and macroalgae, plays a role during photosynthesis and acts as a co-factor in a number of enzymic reactions of crucial importance (Lobban *et al.*, 1985). The effect on Cu^{2+} on marine organisms (see review in Morrisey *et al.*, 1996) and particularly on macroalgae (see review by Gledhill *et al.*, 1997), has been extensively documented. At high concentrations, Cu^{2+} is toxic for most marine organisms; it affects the permeability of the plasmalemma, producing a loss of K^+, which generates changes in the cell volume. At the chloroplast level, Cu^{2+} inhibits photosynthesis and, at large concentrations, it degrades chlorophyll and other photosynthetic pigments (Lobban *et al.*, 1985). Fe^{2+}, essential in the respiratory chain, is not considered as a highly toxic metal, hence there is little evidence of its effects on the structure and organization of marine communities contaminated by mining tailings. Although Fe mining is an extensive activity in northern Chile with significant solid residue discharges resulting from the pelletization process, its effect on marine coastal communities has not been documented. This seems also to be the case at the world level, where the literature refers mostly to the effects of heavy metals on subtidal communities from soft bottoms (e.g. Warwick *et al.*, 1988; Morrisey *et al.*, 1996). As to hard bottoms, data are restricted almost exclusively to the effects of contaminants (organic residues, hydrocarbons and heavy metals) on the individuals (e.g. Munda and Veber, 1996), the populations (Correa *et al.*, 1996) and the communities (Smith and Simpson, 1995) of intertidal environments.

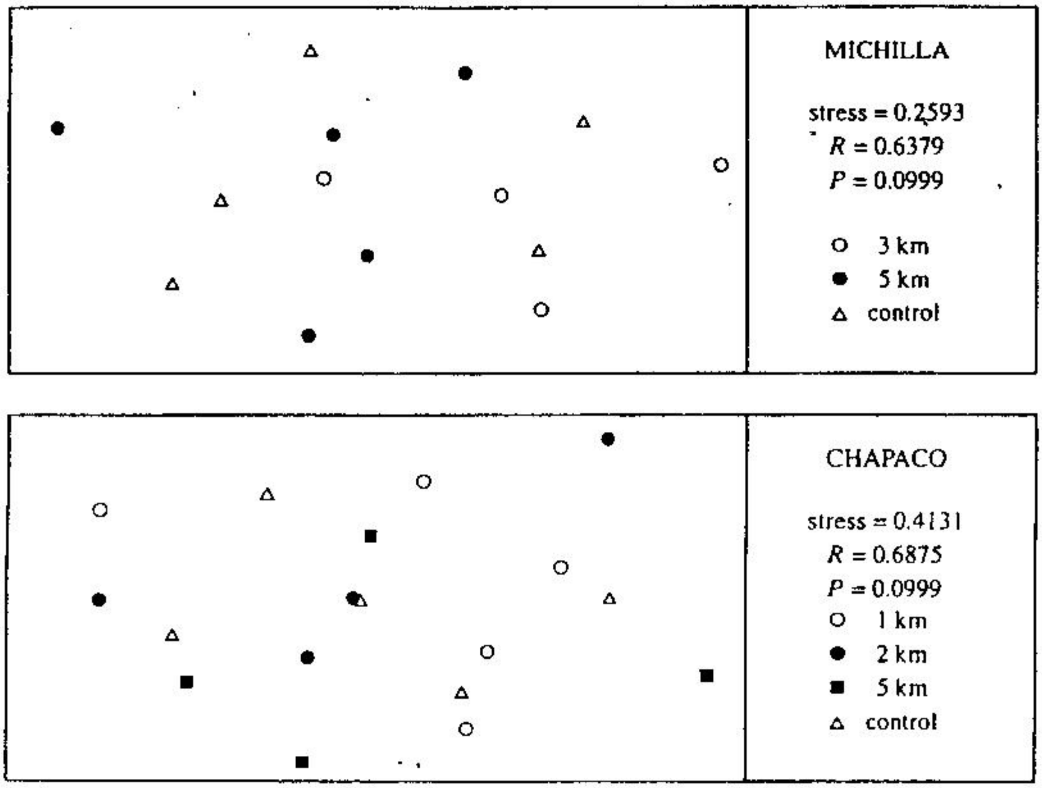


Figure 9 MDS distributions of abundance of fauna associated with *L. trabeculata* from Chapaco and Michilla.

The north of Chile concentrates Fe and Cu mining activities, generating over 60% of the country's total income. This activity, of strategic and socio-economic importance, has affected coastal marine communities over the last 100 years. This work contains the first data on the effects of mining tailings on marine subtidal populations and communities of the South-Eastern Pacific.

We show here that tailings from copper mining cause more ecological perturbations than those from iron mining. Even though the mining activities at Michilla between 1971 and 1994 discharged an annual average of 415 155 tonnes of solids directly to the sea, perturbations are restricted to 0–30 m in depth and up to 3 km distance from the contamination source. The lack of algae and other benthic organisms in a depth gradient, close to the Cu and Fe tailings, could be caused by the accumulation of fine sediments more than by the heavy metal contents *per se*. The 9 200 000 tonnes of sediments dumped into the sea over 19 years at Michilla seem to be the cause of the damage in the subtidal benthic communities of the study area. Similar ecological damage has been documented by Castilla and Nealler (1978) for intertidal communities, as the result of El

Salvador copper mine activities in northern Chile. Resuspension of sediments during winter storms is one important factor that must be considered in the distribution and variability of heavy metal concentrations in seawater and algae. The probability of higher heavy metal concentrations could be minimized by the greater water movement and dilution effects by southwest wind and surge predominant at this time of the year.

Tailings from Fe mining do not generate severe modifications of the subtidal populations and communities in the studied areas. In this sense, a few metres away from the tailings outfall, the *Lessonia* populations occur with similar morphology and abundance as in distant sites and are also similar to the populations of the control localities. No effects on the fauna associated with *Lessonia* holdfasts were detected in the sampling gradient. Though the values of Cu^{2+} and Fe^{2+} exceed the 'normal values' determined for seawater in other latitudes (Lewis, 1994) they do not seem to generate modifications *per se*. This study shows that there are subtidal communities dominated by macroalgae (e.g. *Lessonia*) in areas with high Cu^{2+} and Fe^{2+} concentrations in seawater, as in the case of Carrizal Bajo (28° 05′ S), Caleta Constitución (23° 25′), or those reported by Vásquez and Guerra (1996) for other localities of northern Chile. The effects of sediments that accompany high concentrations of heavy metals from mining tailings seem to produce a greater effect than the toxic cations *per se*. The kinds of physical perturbation that limit the amount of light and maximize abrasion phenomena have not been assessed in the Chilean littoral, and are only scarcely documented in other parts of the world.

The binding of metal ions to polyphenols has been described by Ragan *et al.* (1979) and Pedersen (1984). Karez and Pereira (1995) studied the metal contents in polyphenolic fractions extracted from the brown seaweed *Padina gymnospora* from the southern Brazil coast. They found that concentrations of Zn, Cd, Pd, Cr and Cu were as much as two orders of magnitude greater than in whole plants. In general, Cu and Pb were more concentrated in polyphenols than Zn, Cd and Cr at the most contaminated area. Similar results have been found for the Cu and Fe contents in alginates in comparison with the concentrations of these cations in plants of *L. trabeculata* in the study sites. Our results represent the first reported for the heavy metal contents in alginic acid. Although Paskins-Hurburt *et al.* (1976) reported the metal binding properties of fucoidan, the results for the presence of heavy metals in the alginic acid of Lessoniaceae represent the first report in the literature.

The analysis of Cu^{2+} and Fe^{2+} contents in the different structures of *L. trabeculata*, particularly in alginates, show this macroalgae to be a good indicator of the heavy metal levels in seawater, though they do not disclose clearcut variability patterns. As suggested by Vásquez and Guerra (1996), *Lessonia* spp does not occur in highly contaminated sites (ruderal species) as it is a sensitive species, hence it is not adequate as bioindicator of highly stressed environment situations. Notwithstanding this, its wide distribution range in subtidal communities of the southwest Pacific (Villouta and Santelices, 1986) makes this species a useful tool as study unit, not only because its individual and population characteristics may help to evaluate chemical perturbations within a distribution gradient, but also because their holdfasts contain a rich community of associate macroinvertebrates (Vásquez, 1992). These communities are discrete and biologically delimited, allowing replicability in latitudinal and bathymetric gradients. Moreover, as a function of the perturbation to which they are exposed, the populations occurring inside them respond differentially to the different perturbations according to their own tolerance ranges (Vásquez *et al.*, in press).

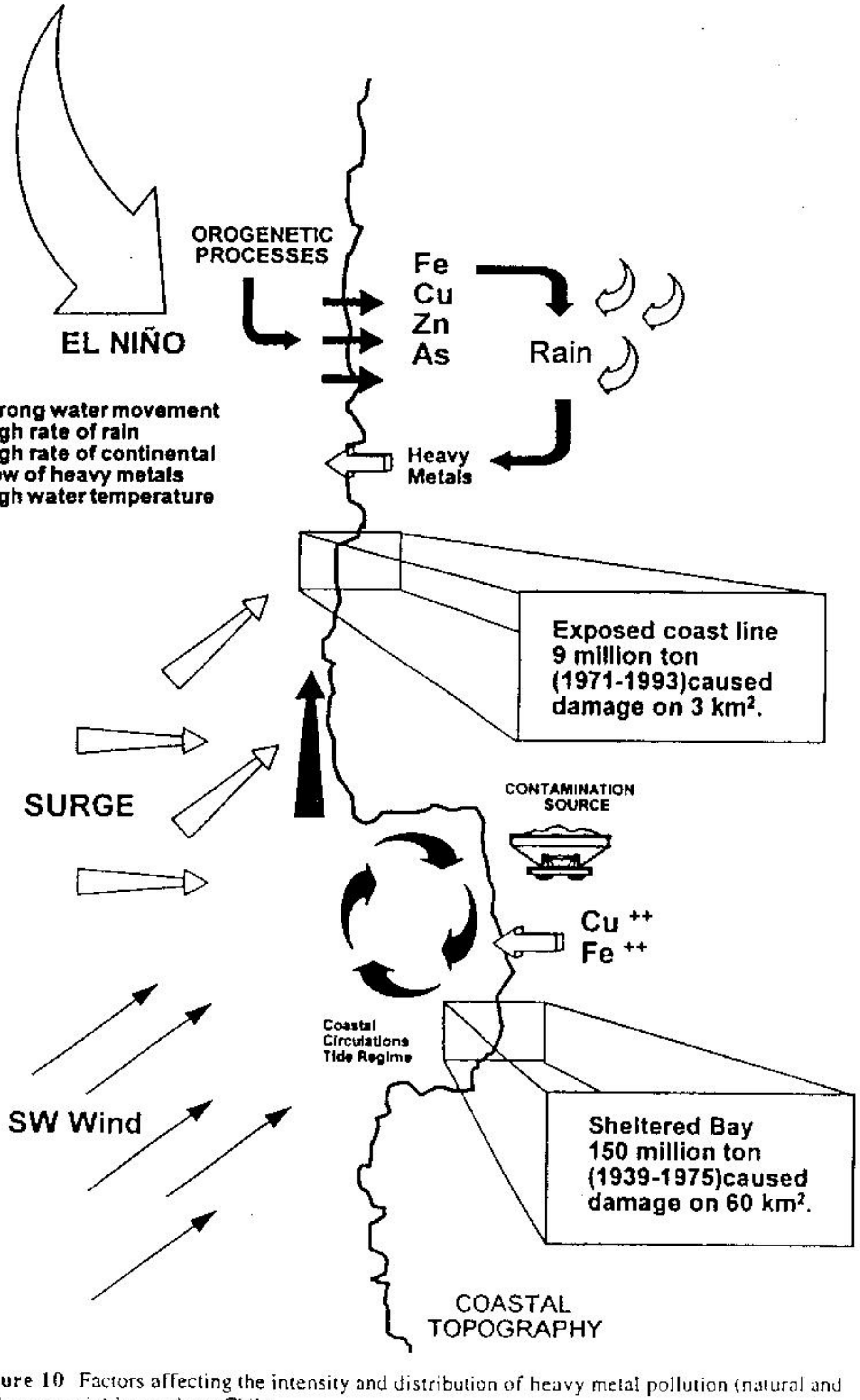


Figure 10 Factors affecting the intensity and distribution of heavy metal pollution (natural and anthropogenic) in northern Chile.

This work shows that the values of heavy metals in seawater, plants and alginates of *L. trabeculata* in contaminated and pristine sites are highly variable. In intertidal areas adjoining the tailing ducts, high values of Cu and Fe are observed, similar to those found in areas with the same contaminant agents in other localities of the Chilean coast (Vásquez and Guerra, 1996; Correa *et al.*, 1996). These high values decrease with depth and with distance from the contamination source. It is important to note that, for concentrations of heavy metals equal to those measured in the study areas, laboratory experiments show non-viability of individuals or a dramatic decrease of the reproductive and growth strategies of experimental individuals (Bryan and Langston, 1992; Anderson and Kautsky, 1996; Gledhill *et al.*, 1997). Environmental factors such as temperature, wind intensity, tidal regimes, water movement, wave impact, coastal circulation, local orogenetic processes, tectonic movements, shore topography, coastal upwellings and global oceaonographic phenomena, such as El Niño, have not been considered in evaluating the effects of contaminant agents on marine coastal communities. All of these environmental factors (Figure 10) must be considered in the interpretation of the effects of heavy metals in field conditions. Future field studies should consider not only the extreme effects of contaminants, but also an intensity gradient that would disclose the mechanisms used by organisms, populations and communities to minimize the effects of these agents of environmental perturbation.

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