Edge effect on a *Neritina virginea* (Neritimorpha, Neritinidae) population in a black mangrove stand (Magnoliopsida, Avicenniaceae: *Avicennia germinans*) in the Southern Caribbean

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Abstract. Mangroves in the Caribbean and particularly in the Urabá Gulf (Colombia) are strongly threatened by selective logging and conversion to pastures and croplands. Specifically, extensive *Avicennia germinans*-basin stands were converted to pastures during the twentieth century, thus exposing benthic fauna to an edge effect. We measured this effect on the population of a numerically dominant gastropod (*Neritina virginea*). Despite its resistance to natural disturbances, it is sensitive to extreme anthropogenic disturbances, and it would therefore be a good biological indicator of basin-mangrove conversion to pastures. Forest structure variables, soil texture, porewater properties and snail density and size were measured in quadrats placed in pastures, pasture-mangrove edges, and mangrove interiors. Snail abundance sharply decreased from the mangrove interior to the edge and then gradually towards the pastures. Individuals in the pasture were predominantly >10mm, and they frequently exhibited shell corrosion compared to individuals found in the interior. There were increases in soil temperature and pH (but oxygen) from interior to pasture consistent with canopy openness. The occurrence of the mangrove edges has led to a marked ecosystem-wide deterioration; however, *N. virginea* (abundance, size, shell corrosion) could be used as a reliable short to midterm indicator of microhabitat and microclimatic differences observed across mangrove-pasture edge.

Key words: basin mangrove, pasture, gastropods, Urabá Gulf

Resumen. Efecto de borde sobre la población de *Neritina virginea* (Neritimorpha, Neritidae) en un rodal de mangle negro (Magnoliopsida, Avicenniaceae: *Avicennia germinans*) en el sur del Caribe. Los manglares del Caribe, particularmente del golfo de Urabá (Colombia) están fuertemente amenazados por la tala selectiva y la conversión a cultivos y potreros. Específicamente, grandes extensiones de rodales de cuenca de *Avicennia germinans* fueron convertidos a potreros durante el siglo veinte, exponiendo así a la fauna béntica a un efecto de borde. Medimos este efecto sobre la población de un gasterópodo numéricamente dominante (*Neritina virginea*). A pesar de su resistencia a las perturbaciones naturales, es sensible a perturbaciones extremas antrópicas, y por lo tanto, puede ser un buen indicador biológico de la conversión del manglar de cuenca a potreros. Variables de la estructura del bosque, textura del suelo, propiedades del agua intersticial y densidad y tamaño del caracol fueron medidas en cuadrantes ubicados en los potreros, en el límite potrero-manglar, y en los interiores del bosque. La abundancia del caracol disminuyó bruscamente desde el interior del manglar hacia el borde y gradualmente hacia el potrero. Los individuos en el potrero fueron predominantemente >10mm, y frecuentemente exhibieron desgaste en la concha, comparados con los encontrados en el interior. Hubo un incremento en la temperatura y el pH del suelo (pero no del oxígeno) desde los interiores hacia los potreros, consistente con la apertura del dosel. La presencia del borde ha llevado a un marcado deterioro del ecosistema, sin embargo, *N. virginea* (abundancia, tamaño, desgaste concha) puede ser usado como un indicador confiable a corto y mediano plazo de las diferencias en microhábitat y microclima observadas a través del límite manglar-potrero.

Palabras clave: manglar de cuenca, potrerización, gasterópodos, golfo de Urabá

Introduction

Anthropogenic edges, as a straightforward consequence of the widespread deforestation and fragmentation (FAO 2011), are key drivers of the worldwide loss of species, especially in the Tropics (e. g. Murcia 1995, Fox *et al*. 1997, McIntyre & Hobbs 1999, Ries et al. 2004, Cavalcanti & Rodal 2010), but no study has focused on mangroves. Anthropogenic edges are formed after forest clearing to establish a different vegetation cover for different uses such as agriculture and cattle ranching, thus changing microclimatic and soil conditions not only at the open land but also several meters inside the untouched forest, and ultimately contributing to decimating or eliminating sensitive populations of plants and animals (Murcia 1995, Ries et al. 2004). Anthropogenic edges are also evident in mangrove forests because this ecosystem is globally one of the most threatened due to deforestation and reclamation (Farnsworth & Ellison 1997, Duke *et al.* 2007, Alongi 2008). Nowadays, although many studies have focused on mapping and quantifying mangrove loss, there are a few comprehensive assessments on the impacts of mangrove deforestation (Blanco *et al.* 2012). However, in contrast to the Old World mangroves, there is little understanding about spatial patterns, rates and ecological impacts of deforestation in the Neotropics (Álvarez-Leon & Polanía 1996, Ellison & Farnsworth 1996). The available studies in African and Asian mangroves (Fondo & Martens 1998, Sasekumar & Chong 1998, Diop *et al.* 2001) have demonstrated that without mangrove exploitation, the tree coverage offers spatio-temporal stability in physicochemical pore-water variables and therefore it weakens the correlations between gastropod microdistribution and those variables. On the contrary, correlations with physicochemical variables effects such as temperature and pH are strengthened in deforested areas (Fondo & Martens 1998). This negative influence of deforestation on epifauna, could be present in Neotropical mangroves, given the high clearing rates documented in some locations (e.g. 0.85%/yr in Quintana Roo, Mexico, Hirales-Cota *et al.* 2010). In these mangroves, forest structure, and therefore epifauna, may respond in a complex way to the anthropogenic forest-gaps and edges as proposed by individual-based models (Berger *et al.* 2008).

Mangroves in the Caribbean coast of Colombia, as most forests in this region (e. g. Etter & Wyngaarden 2000, Gómez *et al.* 2005, Etter *et al.* 2006), are experiencing widespread fragmentation, clearing and conversion to other land covers, and therefore, anthropogenic edges have become a dominant landscape feature. As an example, we recently proposed that the Urabá Gulf, located in the southwestern Colombian Caribbean coast, in the vicinity of the border with Panama, is a deforestation hotspot where coastal floodplain forests and mangroves were decimated during the second half of the twentieth century (Blanco 2009, Blanco *et al.* 2012). Although it is located within a worldwide known biodiversity hotspot (Chocó-Darién Ecoregion), overexploitation of natural coastal resources has increased extensive banana and plantain crops at the expense of the native forests during the last six decades. This situation has decimated and fragmented native forests and created long and marked edges with crops and pastures (Arroyave *et al.* 2012). Mangroves, in particular the basin type, have been encroached by pastures in some sectors of the Eastern Coast of the Urabá Gulf, where the formerly *Rhizophora mangle* (Linnaeus 1753) and *Avicennia germinans* (Linnaeus 1764) –the dominant stands– have been replaced by *Laguncularia racemosa* stands (Gaertner 1807) as a consequence of both, selective over-logging and rapidly expanding cattle ranching, thus causing clearly-defined anthropogenic edges (Corpourabá 2003, Taborda 2008, Blanco *et al.* 2012). The ecological consequences of these edges have not been evaluated, although recent studies suggested a decline in epifauna due to mangrove conversion to pastures (Blanco & Castaño 2012).

Within the above mangrove-pasture transition, it is noteworthy that *Neritina virginea* (Linnaeus 1758) (Neritimorpha: Neritidae) is one of the most abundant benthic macroinvertebrates (García & Palacio 2008, Ortiz & Blanco 2012). It is numerically dominant on the sediments, and on mangrove prop-roots and trunks (Ortiz & Blanco 2012), and it is widespread in the Caribbean, colonizing mangroves and seagrasses in coastal lagoons, bays and gulfs, but also rivers up to several kilometers inland (Blanco & Scatena 2006). In this paper, we used *N. virginea* as possible bio-indicator of mangrove-pasture edge effect because individuals exhibit: a) a significant resistance to natural and anthropogenic disturbances and fluctuations in water level and microhabitat, thus quickly recovering from minor disturbances (Blanco & Scatena 2006, Blanco & Arroyave 2009, Blanco & Castaño 2012), and b) they show a widespread distribution and strong tolerance to wide variations in salinity, pH, dissolved oxygen concentration, and soil

texture (Blanco & Scatena 2006, Blanco & Castaño 2012). These features are shared with other mangrove gastropods reported as bioindicators elsewhere (e.g. Vermeij 1971, Cantera *et al.* 1983). This paper was specifically aimed at determining: 1) how snail density and size vary along the mangrove-pasture edge in the Turbo River Delta (Urabá Gulf), and 2) how mangrove conversion to pastures produces sub-lethal effects such as shell corrosion.

Materials and Methods
Study area
        The Urabá Gulf is located in the Southwestern of the Colombian Caribbean, with an estimated area of 4291km^2, and a 609km-coastline where extensive mangroves are settled (Corpourabá 2003, Blanco *et al.* 2012). We conducted the present study in El Uno Bay,

located in the middle part of the Eastern Coast, where quaternary sedimentary sandflats have promoted the establishment of mangroves (Corpourabá 2003, Taborda 2008). Currently, there are 145ha of mangroves, where red mangrove fringes (*R. mangle*) are well distinguished from inland basin mangroves dominated by black mangrove (*A. germinans)* (Blanco *et al.* 2012) (Figure 1). Unplanned selective logging and conversion of mangroves to pasture lands are responsible for the formation of canopy gaps and anthropogenic edges (Corpourabá 2003, Blanco *et al.* 2012, Blanco & Castaño 2012). Recently published faunal inventories have highlighted the poverty of benthic communities in the region compared with mangroves located in the open Caribbean Coast of Colombia (García & Palacio 2008, Blanco & Castaño 2012, Ortiz & Blanco 2012).
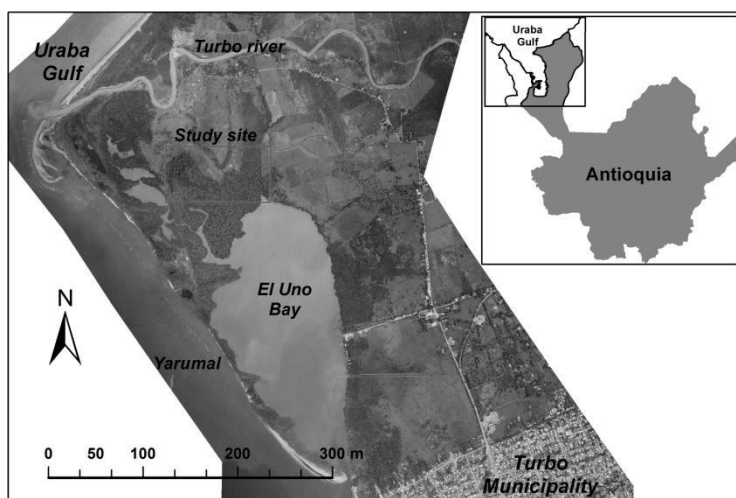

Figure 1. Map of the study site in the Urabá Gulf, Department of Antioquia (Colombia).

Sampling and data analysis
        We carried out a monthly sampling between May and July 2010 because this is the period of highest densities of *N. virginea* (personal observations), and hence the three months might be considered as statistical blocks with relatively low variability, and due to logistics restrictions for conducting a year-long study. A mangrove-pasture edge was divided into three blocks (considered as statistical blocks), located 100m apart one to another, and each one was divided into 9 individual plots of 100m^2, separated 30m from each other, in the following zones: a) pasture (n= 3 plots), outside of the mangrove; b) edge (n= 3 plots), located in the mangrove-pasture edge, one half covered by mangrove and the other half covered by pasture; and

c) interior (n= 3 plots), inside the mangrove basin (Figure 2).

        In order to assess the forest structure, diameter at breast height (DBH), basal area and height were measured for all individuals of *A. germinans* found within the 27 plots of 100m^2. One plot was selected within each zone per block (9 plots for entire area) to record the following environmental variables, within 10 randomly-placed 1m^2-subplots per plot: percentage of canopy openness with a Spherical Crown Densitometer, and the porewater pH, temperature (°C) and oxygen (mg/L) with a YSI multiprobe. The same sampling design was used to assess population variables of *N. virginea.* The total number of individuals present in every 1m^2-subplot was counted to determine abundance, spatial distribution pattern,

absence/presence of shell corrosion (due to erosion and rasping caused by co-specifics to supplement calcium carbonate [Blanco & Arroyave 2009]) and body size frequency distribution (accounting the individuals in the following categories measured with a caliper: 1) ≤10mm, 2) 10mm, 3) 10-15mm, 4) ≥15mm).

Statistical analyses were performed using Minitab 16.0. Normality was tested with Kolgomorov-Smirnov test and data were log-transformed when necessary. Forest structure data were analyzed non-parametrically. Differences in medians of abundance, DBH, and height among blocks and zones were tested using the nonparametric statistical Kruskal-Wallis (H) test.

In the same way, this test was applied to shell corrosion among zones. Data of environmental variables and the population variables (abundance and body size) were altogether analyzed from three sources of variation, a) zone, b) sampling month, and c) interaction, using a balanced Multivariate Analysis of Variance (MANOVA). Afterwards, we used the Wilks and Pillai statistical test to determine the level of significance of the differences within levels of explanatory sources of variation, and we recorded Pearson correlations among response variables to establish relationships. We prefer a multiple response ANOVA over a single response method to test the ecosystem-wide influence of deforestation along anthropogenic edges. A nested design was not selected because no statistical difference was detected among blocks and therefore, the subplots within each zone were considered as replicates. We used the mean/variance ratio to determine if spatial distribution was clumped, uniform or random compared to a standard statistical model (Pielou 1977).
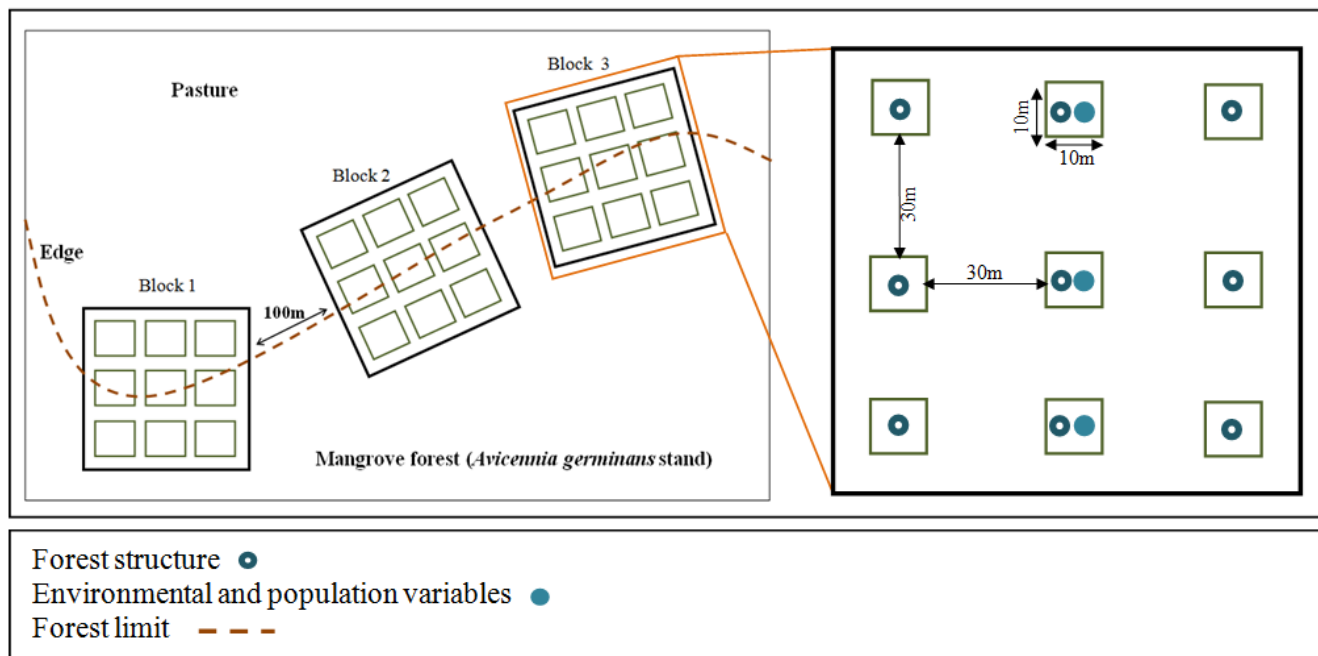

Figure 2. Sample design of the study. The area covered three zones (Pasture, Edge and Mangrove forest) divided into three blocks where forest structure, environmental variables and population variables of *Neritina virginea* were measured into 100m² plots.

Results

Forest structure

The number of *A. germinans* trees was lower in the pasture, and their height and DBH were not the highest, thus suggesting that the oldest trees were cut, and selection of trees left in place was random (Table I). Meanwhile, there were no significant differences in medians of the mangrove forest variables among blocks (Kruskal-Wallis test, p>0,05), therefore they were considered as statistical replicates for further analyses of environmental and population variables. However, there were differences among zones along the mangrove-pasture edge (p<0,001).

Table I. Average values and standard deviations of the variables evaluated for *Avicennia germinans* stand.

	Zone	Abundance (# trees)	DBH (cm)	Height (m)
BLOCK 1	Pasture	2	29,6±26,9	12,4±5,3
	Edge	10	35,5±11,2	8,2±2,5
	Interior	19	36,6±13,5	10,2±1,91
BLOCK 2	Pasture	0	-	-
	Edge	13	38,3±13,6	13,1±3,4
	Interior	16	39,2±10,4	13,9±2,3
BLOCK 3	Pasture	0	-	-
	Edge	18	39,7±8,9	10,7±2,9
	Interior	24	39,1±16,5	11,5±2,8

Environmental variables

The environmental variables showed significant differences according to zone and sampling month (MANOVA: $p<0,001$; F= 68,1; df= 8). The zone x month interaction indicated a significant spatio-temporal variation (MANOVA: $p<0,001$; F= 7,0; df= 16). Meanwhile, porewater temperature was significantly higher in the pasture than in the mangrove interior ($p<0,05$), according to canopy openness (Pearson correlation= 0,614. $p<0,01$), while pH was marginally lower tending to acidity ($<6,5$). Dissolved oxygen concentration was not significantly different among zones, but it varied among sampling months ($p<0,05$), ranging from 4,26 to 0,31mg/L, according to river flooding (Figure 3). No correlations were observed among the rest of environmental variables.
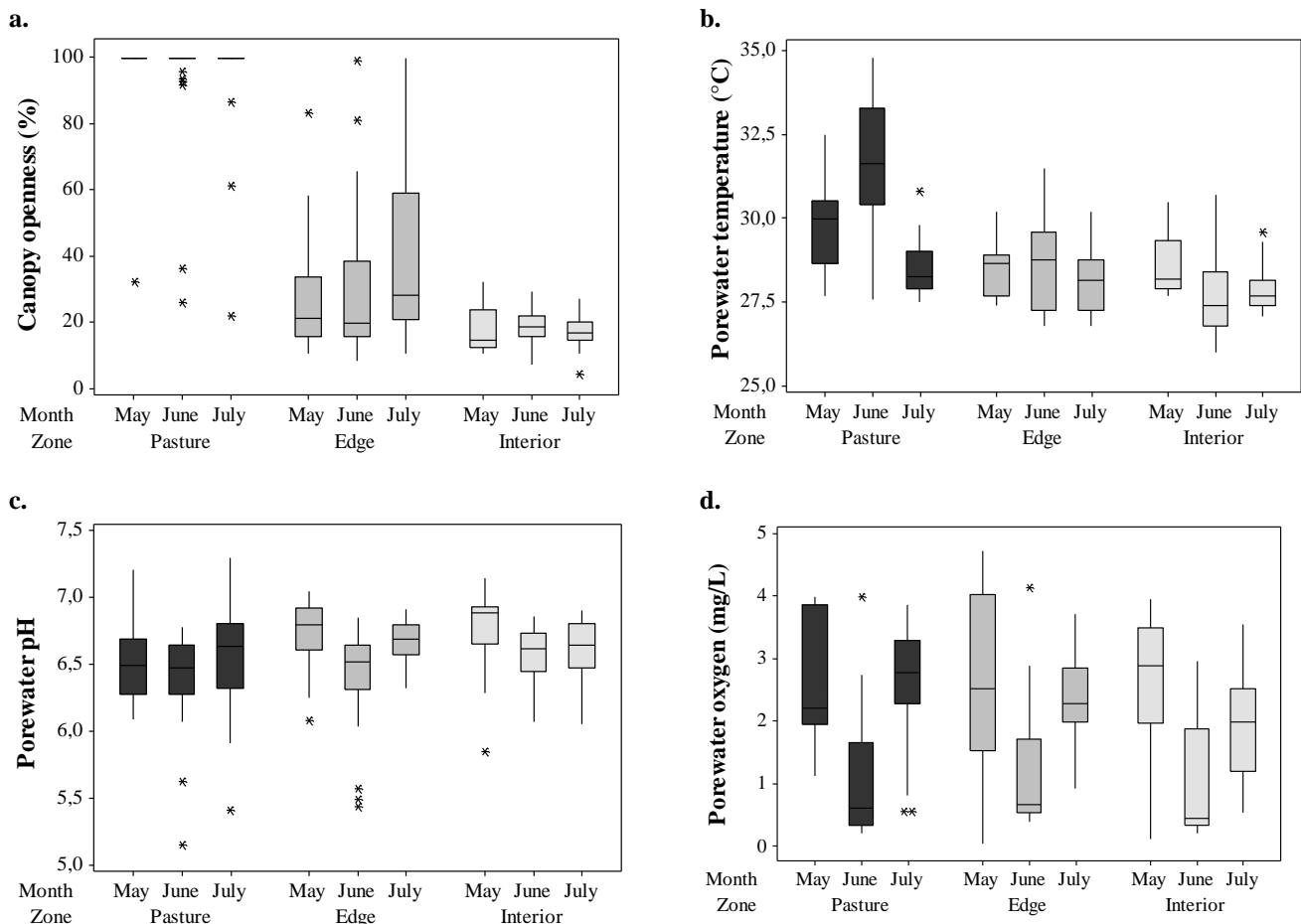

Figure 3. Box and whisker graphs showing data distribution over time (month) and space (zone) n= 270 $\overline{x}$ ±sd : a) percentage of canopy openness ($\overline{x}$ =48,9%±37,7); b) porewater temperature ($\overline{x}$ =28,8°C±1,6), c) porewater pH ($\overline{x}$ =6,5±0,3), and d) porewater dissolved oxygen ($\overline{x}$ =2,0mg/l±1,1).

N. virginea population variables

We recorded a total of 4671 individuals of *N. virginea*, exponentially decreasing in relative abundance from subplots located in the mangrove interior to the edge and to the pasture (87,2%, 11,8% and 0,9%, respectively). Relative abundance gradually increased from May to July (Figure 4). In turn, density and body size varied significantly according to the zone (MANOVA: $p < 0,001$; $F = 54,8$; $df = 6$), the sampling month ($p < 0,001$; $F = 13,69$; $df = 6$) and the interaction ($p < 0,001$; $F = 8,57$; $df = 12$). Additionally, we found weak correlations between density and body size (26-62%), being June the month when we found the biggest individuals. Density was statistically different among zones (ANOVA: $p = 0,008$; $F = 5,42$; $df = 2$), because it was higher in the mangrove interior ($45,6 \pm 52,1 \text{ind/m}^2$) compared to the edge ($6,9 \pm 6,8 \text{ind/m}^2$) and the pasture ($0,8 \pm 1,4 \text{ind/m}^2$), that zones showed no difference between them (Tukey Test). Over the entire area, the spatial distribution pattern of individuals was highly clumped, because the density variance (2415,5) was significantly higher than the density average ($17,4 \text{ind/m}^2$).

Individual shell size ranged between 5 and 18mm, but 95,7% of the individuals were within the 10-15 mm range, and they decreased in number exponentially from the mangrove interior towards the pasture (Figure 5). Individuals <10mm prevailed in the mangrove interior and they were almost absent in the pasture. Individuals >10 mm were more abundant in the interior and the edge than in the pasture, while individuals >15mm were dominant in the pasture. On the other hand, size categories 1 (<10mm) and 3 (10-15mm) showed significant differences in abundance of individuals (MANOVA: $p < 0,001$), and an interaction with the zone was observed ($p < 0,03$), mainly because there were few individuals <10mm on the edge.

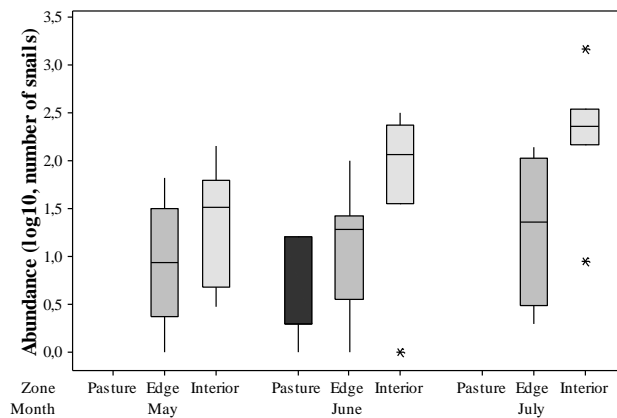**Figure 4**. Abundance of *Neritina virginea* individuals according to zone ($\overline{x} \pm$ sd) (Pasture: $\overline{x} = 6,0 \pm 6,8$. Edge: $\overline{x} = 27,7 \pm 36,7$. Interior: $\overline{x} = 169,8 \pm 299,4$) and sampling month (May: $\overline{x} = 33,6 \pm 41,4$. June: $\overline{x} = 54,0 \pm 86,4$. July: $\overline{x} = 262,0 \pm 421,0$).
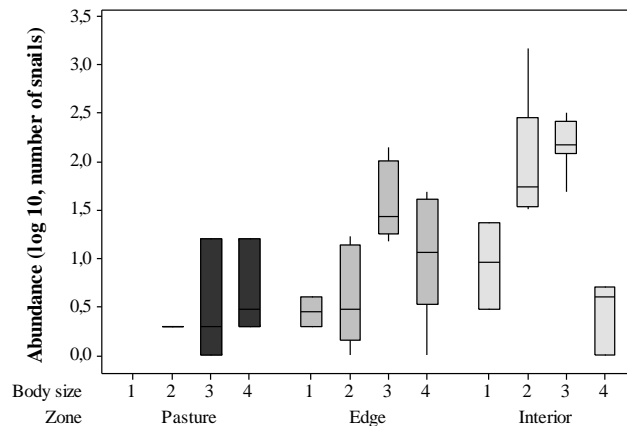

Figure 5. Relative abundance of *Neritina virginea* individuals according to shell size categories ($\overline{x} \pm$ sd): 1) ≤10mm ($\overline{x} = 20,5 \pm 20,5$), 2) 10mm ($\overline{x} = 823 \pm 1394$), 3) 10-15mm ($\overline{x} = 682 \pm 842$), 4) ≥15mm ($\overline{x} = 49 \pm 58,3$). Observe the absence of small individuals in pasture and the presence of all shell size categories in a higher abundance in mangrove interior.

Shell corrosion comprised 20,6% (673 individuals) of total population. The highest proportion of shell-eroded individuals (28,3%) was observed in June, followed by July (11,3%), and the

percentage of eroded shells significantly increased from the mangrove interior towards the pasture (Kruskal-Wallis test p<0,01) (Figure 6).
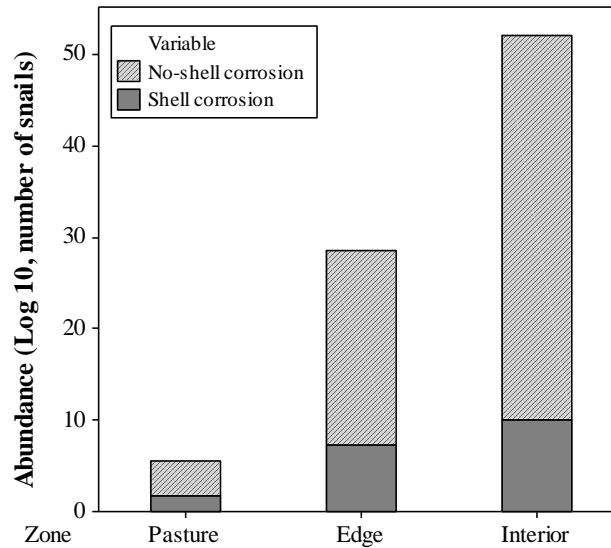

Figure 6. Relationship between shell corrosion and total *Neritina virginea* individuals by zone. Observe in the pasture the increased proportion of eroded shells despite of the low total abundance of individuals.

Discussion

This study demonstrated that mangrove-pasture transitions are responsible for negative edge effects on the abundance and size-distribution of the dominant gastropod *N. virginea*. In this case, the edge effect translated in a 99% loss of tree cover in the pasture and a 34% in the edge, similar to the indicated as a human disturbance in other forest types (Murcia 1995, Berlanga & Ruiz 2007). This anthropogenic edge effect seems to be mediated by changes in biotic and abiotic variables, like increasing insolation that indirectly increases the loss of soil moisture and porewater temperature, and probably acidification in the three zones (pastures, edge and interior), while the soil texture was consistently similar and dominated by fine sands (Taborda 2008). It also has been found that these variables limit the settlement and development of mangrove seedlings and their associated fauna in other studies (Kerr & Deguise 2004, Echeverría *et al.* 2007).

Anthropogenic edges may promote sublethal effects, and in our case shell corrosion is considered as such because weakens the shells and may further favor predation by birds and decapods (Wells 1980, Blanco 2001, Blanco & Arroyave 2009). The high percentage of eroded shells (53%) found in the pasture in

comparison to the mangrove interior (18%) could be related to the lower porewater pH observed. Nearly neutral pH in mangrove porewater is maintained by tidal flushing (Mazda *et al.* 1990), thus a reduction from the mangrove edge to the pasture may be promoted by the following hypothetical combination of factors. In first place, considerable amounts of leaflitter and wood are translocated to the ground after mangrove clear-cut (personal observations), and therefore could promote acidification during its decomposition. Second, formation of temporary pools during rainy season, their high temperatures and subsequent desiccation, could promote further weathering and decomposition of leaflitter. Third, abundant livestock manure was observed near mangrove edge thus contributing to acidification thru decomposition. Water in ponded areas in the pasture is only replenished during heavy rains or during Turbo River overflooding (personal observations). However, in the long term, soil alkalinization seems to occur in pastures, where the original mangrove leaflitter was completely decomposed or washed, soil organic matter was exhausted due to livestock trampling and meteorization, and the sandy-loamy matrix was exposed, as suggested by Blanco & Castaño (2012) for similar pastures located hundreds of meters away from

the mangrove edge in this study. Finally, the occurrence of sublethal effects are supported by the weak relationship between instantaneous water physicochemical and microclimatic variables and snail abundance (r^2 range: 8,2-43,6%). This finding further suggest that *N. virginea* is highly tolerant to the great micro- and macro-scale variations observed in Urabá Gulf mangroves and estuaries, where salinity varies between 0,1 and 25,9 (Ortiz & Blanco 2012, Blanco & Castaño 2012). Therefore, snail populations could be reduced in terms of abundance due to extreme climatic disturbances (e.g. droughts) and habitat modifications induced by mangrove-pasture transitions.

The observed patterns in snail abundance and shell size distribution seem to be explained by increased mortality due to predation related to the altered microhabitat conditions in edges and recently established pastures. In the study mangrove interior, gastropod individuals were found clinging on *A. germinans* pneumatophores, where they climbed during flooding tide (personal observations). Since pneumatophores probably ease snail settlement and feeding, and provide protection against predators (Skilleter & Warren 2000, Echeverría *et al.* 2007), therefore, removal or die-off of pneumatophores may partially explain the scarcity of snails in the pasture and the edge in our study site. Furthermore, the fact that the largest individuals (>15mm) were the only withstanding in pastures established in previous weeks or months (personal observations), suggests that the smallest individuals died due to either a greater vulnerability to extreme environmental conditions or a greater predation risk. Several studies in Puerto Rico indicated that predation risk is greater in small individuals of *N. virginea* and it is reduced when they reach a size that confers a refuge from predatory decapods (Blanco & Scatena 2006, 2007, Blanco & Arroyave 2009). In contrast, we found individuals in all body size categories in the mangrove interior.

In addition, the observed patterns in snail abundance and shell size distribution also seem to be explained by a disrupted individual migration. The sharp increase in the relative abundance of individuals >10mm and therefore, a net increase in mean shell size population, from the mangrove interior to the pasture, could be due to the migration and growth of individuals along the intertidal zone, similar to the observed along coastal streams in the Caribbean (Blanco & Scatena 2005, 2007). A rheotactic behavior was actually observed during flooding and ebbing tides, and massive clumps were also observed during

the rainiest month (July). The largest individuals (>15mm) in the pasture could be a legacy of past massive migrations before mangrove reclamation, while a lack of small individuals (<10mm) suggested that either hydrological connectivity was lost, or a mass mortality of migrating individuals occurred. However, we did not observe empty small shells, as elsewhere, to support the second hypothesis and hence the first one seems more likely.

Finally, the prevalence of individual clumps of *N. virginea* could be used as a measure of population condition. It is worth noting that the individuals exhibited a highly clumped distribution in all study blocks, although it was more marked in the mangrove interior, a condition consistently documented in many neritids (Blanco & Scatena 2005, Tan & Clements 2008) and related to substrate type, water depth, freshwater discharges and ocean connectivity (Blanco & Scatena 2006). Gregariousness is also hypothesized as an evolutionary strategy to avoid predators, and as a mechanism for trail following during massive upstream migrations (Blanco & Scatena 2005, 2007). Therefore, the scarcity of individual clumps could be a reflection of altered conditions related to land cover and use change due to agriculture and urbanization. Accordingly, the presence or absence of this species in sampling quadrats could be a good alternative for monitoring programs instead of the time-consuming individual counting to estimate population density and, ultimately, the degree of mangrove conservation. We observed a greater number of empty quadrants in the edge and the pasture than in the mangrove interior, supporting previous reports in mangrove gaps and pastures in the same delta (Blanco & Castaño 2012, Blanco *et al.* 2012).

In conclusion, pasture expansion in El Uno Bay has produced an edge effect that, in addition to the evident mangrove-tree clear-cut, has changed microhabitat and surface soil properties, and has directly and indirectly modified the abundance and body size distribution of the dominant detritivore *N. virginea*. Nevertheless, this species demonstrated being somewhat resistant to the high soil temperatures and low pH promoted by mangrove clearing, thus highlighting not only its usefulness as a biological indicator of ecosystem conservation status, but also as a reference species for restoration efforts. We suggest that future studies in Caribbean mangroves should take advantage of the high abundance of neritid snails for addressing both, basic-ecological as well as applied conservation-oriented questions. We also

suggest that this group could be used as a harbinger of ecosystem-wide deterioration beyond natural variability, because any change in a species' distribution may have cascading effects throughout the community (Ries *et al.* 2004). Since *N. virginea* is the dominant benthic consumer in terms of both abundance and biomass in Urabá Gulf mangroves (Ortiz & Blanco 2012, Blanco & Castaño 2012), we propose considering the species as an ecosystem engineer, similarly to crabs elsewhere (Lee 2008). Estuarine and freshwater neritid snails are deposit feeders and facultative detritivores, seemly processing significant amounts of surface sediments and leaflitter (Blanco & Castaño 2012), and therefore their local extinction could bring noticeable negative effects on mangrove benthic functioning. Therefore, in the Urabá Gulf, as in the Greater Caribbean (Alongi 2008), the functional mangrove stability may greatly rely upon the conservation status of the dominant gastropod *N. virginea*.

Acknowledgments

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