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1 Influence of species and stand position on isotopic and molecular composition
2 of leaf litter during degradation in an urban mangrove forest

3 Sarah Louise Robin^{a*}, Claude Le Milbeau^b, Kapeliele Gututauava^a, Cyril Marchand^a

4 ^a Institut de Sciences Exactes et Appliquées (ISEA EA7484), Université de la Nouvelle-Calédonie,
5 145 Avenue James Cook, Nouville, BP R4 98851, Nouméa Cedex, New Caledonia

6 ^b Institut des Sciences de la Terre d'Orléans (ISTO), UMR 7327, Université d'Orléans, CNRS,
7 BRGM, 45071 Orléans, France

8 *corresponding author: sarah.robin@unc.nc

9 **Abstract**

10 Understanding the diagenetic processes of organic matter (OM) within mangrove forests is essential
11 to grasp ecosystem dynamics and exchanges with adjacent ecosystems. This study investigated
12 the influence of urban runoff on leaf litter degradation in a New Caledonian mangrove forest,
13 subjected to urban rainwater for over 50 years. Focusing on *Avicennia marina* and *Rhizophora*
14 *stylosa*, our objectives were to determine factors affecting leaf litter degradation, assess urban runoff
15 effects on decay rates and element concentrations, and understand OM changes at the molecular
16 level. Litterbags containing senescent leaves placed on the mangrove floor were collected after 7,
17 14, 28, 56, and 72 days. C and N along with their stable isotopes were evaluated during degradation
18 as well as lignin and neutral carbohydrates contents. Despite lower initial N content, *R. stylosa*
19 exhibited faster degradation ($t_{1/2}$ of 36 ± 3 and 28 ± 2 days) than *A. marina* ($t_{1/2}$ of 43 ± 9 and 33 ± 4 days),
20 emphasizing the critical role of species position within the mangrove forest regarding tidal
21 immersion. Urban runoff, submerging the urban site, intensified leaf litter degradation, influencing
22 both mass loss and molecular changes in OM. After 72 days of leaf litter degradation, the loss of
23 rhamnose and glucose was more pronounced at the control site (13% and 75% for rhamnose and
24 46% and 53% for glucose for *A. marina* and *R. stylosa*, respectively) compared to the urban site.
25 The study also exposed molecular tendencies during mangrove leaf litter degradation. The content
26 of ferulic acid in the leaf litter decreased at all stands after 72 days of degradation (between -25%
27 and -58%), while total syringyl phenols increased (between +64% and +232%). This research
28 exposed the global implications of urban runoff, indicating accelerated leaf litter degradation in

29 mangrove forests, potentially disrupting C sequestration dynamics and threatening ecosystem
30 services.

31 Keywords: mangrove, litterbag, stable isotopes, lignin, neutral carbohydrates

32 **1. Introduction**

33 Mangrove forests are coastal ecosystems established in tropical and sub-tropical areas
34 around the world, with some species present in temperate regions (e.g. Auckland, NZ). These
35 forests have a very large organic carbon (OC) stock, much larger than any marine and terrestrial
36 ecosystems within the same latitudes (Alongi, 2014). Most of this OC is stored in the soil, but about
37 15% is in the aboveground biomass (Alongi, 2020). 31% of the total net primary production of
38 mangrove forests comes from the production of litter (Bouillon et al., 2008). Once this biomass turns
39 into litter, it becomes a crucial part of the trophic chain of the mangrove forest and adjacent
40 ecosystems. Litterfall degradation actively contributes to the energy and material flow in the forest
41 and is considered an energy source for coastal productivity (Lugo and Snedaker, 1974; Twilley et
42 al., 1986). The litterfall organic matter (OM) can be consumed in the forest floor by autochthonous
43 fauna and microorganisms, dissolved during decomposition, and exported to adjacent ecosystems
44 at ebb tide as debris, dissolved OM, or dissolved inorganic matter, which also has a key role in the
45 trophic chain (Fleming and Lin, 1990; Rezende et al., 1990; Wafar et al., 1997; Kristensen et al.,
46 2017; Kida et al., 2019).

47 Mangrove leaves account for the major part of the litterfall (Twilley et al., 1986; Day et al.,
48 1987), varying between 40 and 90% (Mackey and Smail, 1996). Fruits, flowers, and the woody
49 tissues account for the remaining litterfall. Therefore, most studies on litterfall degradation focused
50 on leaf litter, using the litterbag experiment for comparison with literature (Nordhaus et al., 2017;
51 Yang et al., 2018; Vinh et al., 2020). Leaf litter degradation occurs in three main stages: 1) the initial
52 leaching of soluble compounds, 2) the colonization by microorganisms (i.e., fungi, bacteria, and
53 protozoa) and decomposition via litter breakdown (Fell et al., 1975; Cundell et al., 1979; Fell and
54 Master, 1980), and 3) the slow refractory phase where the least labile molecules are consumed
55 (Valiela et al., 1984; Wilson et al., 1986; Tam et al., 1990). The rate of degradation of the leaf litter
56 depends on multiple parameters, with varying significance depending on the context. The initial

57 chemical composition of the leaves, especially C/N ratio, tannin, and lignin content, is one of the key
58 parameters influencing the decay rates (Robertson, 1988; Tam et al., 1990; Mfilinge et al., 2002;
59 Vinh et al., 2020). The leaf-consuming fauna and microbial communities of the forest floor also
60 impact degradation rates (Fell and Master, 1980; Robertson, 1986; Imgraben and Dittmann, 2008).
61 Many abiotic factors can influence litterfall degradation such as particle size of the forest soils, redox
62 conditions, temperature, available oxygen, and the degree and frequency of tidal inundation (Lugo
63 and Snedaker, 1974; Steinke and Ward, 1987; Chale, 1993; Vinh et al., 2020).

64 Leaf litter degradation in mangrove forests is a source of nutrients; therefore, the change in
65 element contents such as C, N, macronutrients, and neutral carbohydrates during degradation is of
66 interest. The capacity of elements to be mobilised from degrading litter depends on the
67 concentrations of these elements in the soil and incoming water, as well as on their role for litter
68 feeders and whether they form complexes with leachable or refractory compounds (Mfilinge et al.,
69 2002; Hossain et al., 2014; Gautam et al., 2016; Nordhaus et al., 2017). Neutral carbohydrates and
70 organic N-rich compounds are easily degraded from the plant material, while lignin-rich litter resists
71 decay (Fell et al., 1975; Cundell et al., 1979; Twilley et al., 1986; Chale, 1993). The depletion of
72 elements during leaf litter degradation indicates leaching to the surroundings, or breakdown and
73 consumption by microorganisms (Fell and Master, 1980; Fourqurean and Schrlau, 2003; Gautam et
74 al., 2016; Nordhaus et al., 2017). The enrichment of elements and macronutrients in leaf litter during
75 the degradation process is attributed to the production of these elements from microorganisms
76 during decomposition such as the polysaccharides exudates from bacteria (Fell et al., 1975;
77 Robertson, 1988; Mfilinge et al., 2002; Yang et al., 2018), or inputs from the water column (Rice and
78 Windom, 1982; Lacerda et al., 1988; Tam et al., 1990; Vinh et al., 2020). The study of these various
79 elements and molecular compounds such as neutral carbohydrates and lignin give essential
80 information on the litterfall degradation and preservations processes, which directly affect the soil
81 OM and exported OM quality.

82 One of the major worldwide threats to mangrove ecosystems is the urbanization of the littoral
83 zone. Almost half of the world's mangrove forests are located in only 4 countries: Indonesia (28.4%),
84 Brazil (9.4%), Malaysia (5.8%), and Papua New Guinea (5.1%) (Hamilton and Casey, 2016). On

85 average, a third of the population of these countries lives on the coastline (CIESIN, 2012).
86 Consequently, there is competition for space between mangrove ecosystems and economic and
87 urban development. Urbanization can inhibit mangrove landward migration, prevent the water flow
88 into the mangrove forest, and be a source of urban water runoffs, which may contain contaminants
89 (Friess et al., 2019). The inputs of urban runoff can modify the immersion conditions of the mangrove
90 trees, change the physico-chemical parameters of the mangrove soil such as pH, redox conditions,
91 and salinity, and be a source of dissolved nutrients, OM, and multiple contaminants (Cavalcante et
92 al., 2009; Bastakoti et al., 2019; Robin et al., 2022; Fiard et al., 2022). Therefore, urban runoff in
93 mangrove forests may influence the decay rates of the litterfall on the soil surface as well as the
94 quality of this litterfall during degradation. To our knowledge, only one study looked at the effect of
95 anthropogenic sewage on mangrove litterfall degradation and concluded that the discharge had no
96 effect on the degradation processes (Tam et al., 1998). It is therefore essential to collect data and
97 gain knowledge on which mangrove characteristic changes driven by the urbanization of the coasts
98 influence the litterfall degradation processes.

99 New Caledonia is a French archipelago located in the South Pacific Ocean between 20°S
100 and 23°S. Semi-arid mangrove forests cover 80% of the West coast of the main island, which has
101 semi-diurnal tidal cycles (Douillet, 2001). 24 mangrove species develop mainly in monospecific
102 stands, but two are dominant, *Rhizophora* spp., which are present on the seaward side of the forests
103 and represent more than 50% of mangrove plants, and *Avicennia marina*, which are established at
104 higher elevation and on saltier soil, representing about 15% of mangrove plants in the archipelago
105 (Deborde et al., 2015). The population density in New Caledonia is low (16 people per km²), but
106 60% of the population is concentrated in the capital and the four cities nearby, that are all located
107 on the littoral zone where mangrove forests develop (Insee, 2020). Urbanization in New Caledonia
108 has caused the destruction of mangrove forests, but only few studies have been conducted on the
109 influence of urbanization on mangrove forests dynamics (Robin et al., 2022).

110 The study of change in element concentrations during litterfall degradation in a mangrove
111 forest is crucial to understand the OM diagenetic and preservation processes within the forest and
112 exchanges with adjacent ecosystems. The aim of this paper is to evaluate leaf litter degradation

113 dynamics in an urban semi-arid mangrove forest (New Caledonia), receiving urban rainwater for
114 more than 50 years. The objectives of this study are 1) to determine factors influencing the
115 differences in leaf litter degradation between mangrove species (i.e., initial composition of the leaves
116 or position of the stand in the mangrove forest), 2) to assess the effects of urban rainwater runoff on
117 leaf litter decay rates and leaf litter changes in element concentrations, and 3) to understand OM
118 litterfall changes with degradation at the molecular level (neutral carbohydrates and lignin content).
119 We hypothesize that the main factors affecting leaf litter decay rates are the stands position in the
120 intertidal zone and the urban rainwater runoff as higher stands immersion times enhance leaching
121 and urban runoff modifies microbial communities. We also hypothesize that the change in
122 composition of leaf litter during degradation will be mainly influenced by the initial composition of the
123 litter and the redox conditions, both factors dependent on the mangrove species and the urban
124 runoff. To this aim, a litterbag experiment has been undertaken for 72 days in a control and an urban
125 mangrove forest for the two main mangrove species present in New Caledonia, *Avicennia marina*
126 and *Rhizophora stylosa*. Decay rates have been calculated from the mass loss, while total C, total
127 N, and their stable isotopes have been determined for each sample, as well as their neutral
128 carbohydrates and lignin contents.

129 **2. Materials and Methods**

130 *2.1. Study sites*

131 One of the main cities of New Caledonia, Dumbea, has seen its population increase by 33%
132 this last decade resulting in the development of urban infrastructures in the littoral zone (Insee,
133 2020). Two mangrove forests in Dumbea have been chosen as study sites in order to compare
134 forests that share a common geological unit as the source of their watersheds and the same
135 vegetation type. The first site is the control mangrove forest (22°12'08"S, 166°26'20"E), located in
136 the Apogoti Bay. Even though housing lots have developed recently around the mangrove, no direct
137 anthropogenic inputs are discharged in the mangrove forest. The second site is the urban mangrove
138 forest (22°12'39"S, 166°27'19"E). Urban rainwater runoff from the upper allotment (mainly housing
139 lots for 10 000 inhabitants) flows into the mangrove forest and has done so for more than 50 years
140 (Fig. 1). At both sites, the two main mangrove species observed, developing in monospecific stands,

141 are *R. stylosa* and *A. marina*. Visible impacts of urban runoff in the urban mangrove forest are the
142 position of *R. stylosa* stand landward of *A. marina* at the entrance of the runoff, with much higher *R.*
143 *stylosa* trees (about 8-10 m high compared to 2 m high at the control site).

144 2.2. Sampling and processing

145 Yellow senescent leaves about to fall on *A. marina* and *R. stylosa* trees were collected at the
146 control site. To limit factors other than the urban runoff, potentially affecting leaf litter degradation
147 such as the senescent leaves initial composition, it was chosen to use the same senescent leaves
148 samples at both sites. This is why senescent leaves were only collected at the control site. Fresh
149 green leaves were also collected from the trees in triplicates, but at both sites, to determine whether
150 the composition of the leaves change significantly between the fresh and senescing stage. One
151 sample of fresh leaves correspond to 30 leaves picked from 5 different adjacent trees. The leaves
152 were carefully washed and air-dried for 24h at the laboratory. Three random samples of about 50 g
153 of whole senescent leaves per species were frozen, freeze-dried, and ground with a cutting mill,
154 corresponding to the samples at t0 "senescent leaves". Samples of about 6 g of dried *A. marina*
155 whole leaves and 15 g of dried *R. stylosa* whole leaves were prepared and precisely weighed. Each
156 samples were placed into individual nylon litterbags (2 mm mesh size) for the litterbag experiment.
157 The 2 mm mesh size allowed for soil contact enabling bacterial activity while limiting the physical
158 actions of large detritivores such as crabs. A total of 90 bags were attached to the roots of the
159 corresponding tree species, on the surface of the soil, and in three different spots within the same
160 species stand in both the control and urban mangrove sites. Triplicates of litterbags were collected
161 at the three spots, for each species, after 7, 14, 28, 56, and 72 days (a total of 36 bags per day of
162 collection). The experiment was stopped after 72 days because the degradation was already very
163 advanced, and samples further degraded would be too hard to analyse properly. The litterbags were
164 immediately opened at the laboratory, the leaves were carefully washed and air-dried for 24h prior
165 to weighing. Samples were then frozen, freeze-dried, and ground with a cutting mill. Fresh green
166 leaves were also collected immediately from the trees at both sites in triplicates, washed, freeze-
167 dried, and ground.

168 2.3. Mass loss and half-life

169 The percentage mass loss or percentage mass remaining after 7, 14, 28, 56, and 72 days of
170 degradation were obtained with the initial mass (± 0.001 g) of the air-dried leaves prior to placing in
171 the litterbags and the mass of the same air-dried leaves after collection (± 0.001 g) in the mangrove
172 site. For comparison with literature, single exponential trends were plotted for each species:

$$173 \quad W_t = W_0 e^{-kt} \quad (1)$$

174 with W_t the mean mass remaining in g, W_0 the initial mass in g, k the exponential factor, and t the
175 time in days. The half-lives ($t_{1/2}$) in days were obtained with the following equation and the
176 uncertainties calculated with R from the decay rates and their standard deviations:

$$177 \quad t_{1/2} = \ln(2)/k \quad (2)$$

178 *2.4. Carbon, nitrogen, and stable isotopes analysis*

179 Total C, total N, and their stable isotopes were measured on an isotope ratio mass
180 spectrometer Sercon Integra 2 at the LAMA laboratory of the Institute of Research and Development
181 of New Caledonia. About 30 mg of samples were weighed in tin cups and folded into tightly closed
182 spheres. The samples were transported in a combustion column with helium gas where they were
183 heated at 1 000 °C and oxidized with Cu and Cr oxides, while silver wool trapped sulphur and
184 halogens. The samples were then transferred to the reduction column at 600 °C where the Cu
185 interacts with excess oxygen so that the NO_x is reduced to N₂. The water in the samples was trapped
186 with Mg(ClO₄)₂ and N₂ and CO₂ of the samples were separated by gas chromatography. The
187 separated molecules were quantified by mass spectrometry in positive ESI mode. The mass
188 spectrometer measures the relative abundances of isotopes in the sample gas. For carbon, it
189 measures the ratio of ¹³C to ¹²C (expressed as $\delta^{13}\text{C}$), while for nitrogen, it measures the ratio of ¹⁵N
190 to ¹⁴N (expressed as $\delta^{15}\text{N}$). To calculate the δ values, the measured isotopic ratio is compared to a
191 known standard. The standard used is the Vienna Pee Dee Belemnite (VPDB) standard for C, and
192 the atmospheric nitrogen standard for N. For quality control, duplicate of samples were passed
193 through the spectrometer (1 every 10 samples) (Table S1).

194 *2.5. Neutral carbohydrates*

195 Neutral carbohydrates were analysed at the Earth Sciences Institute of Orleans (ISTO).
196 About 50 mg of samples were hydrolysed 4h at 105 °C in pyrex tubes with 5 mL of 4M of
197 trifluoroacetic acid (TFA). After cooling in an ice bath, 500 µL of internal standard (2-deoxyglucose
198 at 8 mg L⁻¹) were added and the sample was centrifuged for 30 min at 5 600 rpm. The extract was
199 evaporated, and the dried samples were kept frozen. Prior to analysis, neutral carbohydrates were
200 derivatized with the addition of 500 µL of ethoxy ammonium chloride solution in pyridine (40 mg mL⁻¹)
201 heated for 1h at 70 °C. 250 µL of BSTFA were added and the sample was once again heated at
202 70 °C for 1h. After evaporation, the residue was dissolved in 2 mL of heptane, and the solution
203 passed through a 45 µm filter prior to injection in the GC-MS. The GC is a Trace 1 300 (Thermo
204 Scientific) coupled to a MS ISQ7000 (Thermo Scientific). The GC is equipped with a Restek Rxi-
205 5Sil MS column (60 m x 0.25 mm, 0.25 µm diameter) with 5 m of pre-column. Helium gas is used
206 as vector and the analysis is carried out at constant 1 mL min⁻¹ flow rate. The injector was heated
207 at 280 °C and the transfer line at 300 °C. 1 µL of sample was injected in splitless mode and passed
208 through the column with a temperature program (Table S2). For the MS analysis the source was
209 heated at 240 °C and the ions obtained via electronic impact (70 eV). The spectra were obtained
210 between 50 and 750 Da with a scan time of 0.4 s scan⁻¹. Quality control consisted of duplicate
211 technical samples (≤11% standard deviation) and measurement of the internal standard (Table S3).

212 2.6. Lignin-derived phenols

213 200 mg of samples were hydrolysed for 3 h at 170 °C in 8 mL of 2N NaOH with 100 mg of
214 Fe(NH₄)₂(SO₄)₂ under N₂. After cooling, the extract was collected once the sample was centrifuged
215 at 3 000 rpm for 10 min. The extract was acidified at pH 1 with HCl and lignin-derived phenols were
216 extracted twice with 3 mL of anhydrous diethyl ether. After complete evaporation, lignin-derived
217 phenols were diluted in 1 mL of 50:50 methanol:H₂O, filtered at 45 µm and analysed via UPLC-MS.
218 Chromatographic separation was achieved using a Vanquish Flex system (Thermo Fisher Scientific,
219 France) consisting of a binary pump, an autosampler and a heated column compartment. Samples
220 were separated on a Hypersil GOLD column (1.9 µm particle size), 50 x 2.1 mm (Thermo Scientific).
221 For elution, a binary gradient program was used (Table S4), the mobile phase was a mixture of
222 ultrapure water containing 0.1% of formic acid (Carlo Erba) (eluent A) and methanol OPTIMA LC/MS

223 (Fischer) with 0.1% of formic acid (eluent B). The flow rate was set to 0.5 mL min⁻¹ and injection (2
224 μ L) was made automatically by the autosampler. UV detection was carried out at 280 nm. MS
225 detection was performed using an Orbitrap Exploris 120 (Thermo Fisher Scientific, France) and
226 operated with the heated electrospray ionization source in negative ion mode with a vaporizer
227 temperature of 300 °C. Scan modes were both full MS – ddMS² and SRM. The full MS mode had a
228 resolution of 30 000 and a scan range of 50 – 250 m/z with HCD Collision energies specified for
229 each molecule. The SRM mode had a resolution of 120 000 at 20% HCD collision energy. The full
230 UPLC-MS optimized parameters are displayed in Table S4. The quantification was performed with
231 an external calibration curve (Table S5) and laboratory reference materials spiked with the analytes.
232 For quality control, a blank solution was passed at the beginning and end of every series in the same
233 conditions as the samples and values were all below detection limits (Table S6). A quality control
234 solution (QC) consisting of a point of the calibration curve was passed every 10 samples in the same
235 conditions as the samples and the values obtained were all below 15% difference with the expected
236 value (Table S6).

237 2.7. Statistical analyses

238 Statistical analyses were performed using R studio software (version 1.2.5001). For
239 comparison between species or study sites the Mann-Whitney test was performed. All other
240 statistical analyses with more than two variables were tested with a Kruskal-Wallis test followed by
241 a Wilcoxon test. All tests were performed with a 95% confidence interval and $n \geq 3$. Kendall
242 correlation analysis was performed to obtain correlation matrices.

243 3. Results

244 3.1. Sources characteristics

245 All measured variables for the fresh and senescent leaves of *A. marina* and *R. stylosa* at
246 both sites are displayed in Table 1. The fresh leaves of both species have significantly higher $\delta^{15}\text{N}$
247 values ($p < 0.01$) and lower $\delta^{13}\text{C}$ values ($p < 0.05$) at the urban site (5.8‰ and -27.7‰, and 2.6‰
248 and -30.7‰, for *A. marina* and *R. stylosa*, respectively) than those collected at the control site (1.1‰
249 and -25.8‰, and -1.3‰ and -27.5‰, for *A. marina* and *R. stylosa*, respectively). At the control site,

250 fresh leaves are more enriched in ^{13}C than senescent leaves for both species, but for *R. stylosa* they
251 are less enriched in ^{15}N (-1.3‰ vs -0.2‰). Both fresh and senescent leaves of *A. marina* have
252 significantly higher N contents ($p < 0.01$), $\delta^{13}\text{C}$ ($p < 0.05$), and $\delta^{15}\text{N}$ values ($p < 0.05$) than *R. stylosa*
253 but lower C/N ratios ($p < 0.01$) (39 for *A. marina* and 166 for *R. stylosa*).

254 In Table 1 it is shown that the total lignin contents ($\Sigma 11$ phenols) of the fresh and senescent
255 leaves of *A. marina* are significantly higher than those of *R. stylosa* ($p < 0.05$), which is also the case
256 for the 4 phenol groups (p-hydroxyl or P, vanillyl or V, syringyl or S, and cinnamyl or C). The
257 senescent leaves at the control site have significantly higher total lignin concentrations than the
258 fresh leaves ($p < 0.01$) by a factor of 10 for *A. marina* and a factor of 6 for *R. stylosa* due to higher
259 vanillyl and cinnamyl contents. Therefore, the ratios $P/(V+S)$, P/V , and S/V are much lower in the
260 senescent leaves than in the fresh leaves for both species. The acid over aldehyde ratios (Ad/Al) in
261 senescent leaves are also much higher than in the fresh leaves (Table 1).

262 Fresh leaves of *A. marina* are more enriched in neutral carbohydrates than *R. stylosa* but
263 the leaves of *R. stylosa* have higher concentrations of hexoses (mannose and galactose) and deoxy
264 sugars (fucose and rhamnose), especially fucose and galactose than *A. marina* (Table 1). Total
265 carbohydrates content is significantly higher in the senescent leaves than the fresh leaves at the
266 control site ($p < 0.05$) (Table 1). Neutral carbohydrates represent 45.2% and 49.9% of the total mass
267 for the senescent leaves of *A. marina* and *R. stylosa*, respectively (Table 1). Xylose, arabinose, and
268 glucose are the most concentrated neutral carbohydrates measured representing up to 33%, 34%,
269 and 42 % of the total neutral carbohydrates, respectively. Ribose and fucose are the least
270 concentrated neutral carbohydrates measured in both fresh and senescent leaves with values as
271 low as 0.48 mg g^{-1} , and 0.71 mg g^{-1} , respectively (Table 1).

272 3.2. Decay rates

273 The decay rates of *A. marina* and *R. stylosa* leaf litter degradation at both sites are displayed
274 in Figure 2. The calculated half-lives are 43 ± 9 days and 36 ± 3 days for *A. marina* and *R. stylosa* at
275 the control site, respectively, and 33 ± 4 days and 28 ± 2 days at the urban site, respectively. At both
276 sites, in the first 7 days, *A. marina* leaves lost 41% of their initial mass and *R. stylosa* leaves lost

277 61% of their initial mass. After 72 days, on average, 26% and 19% of the initial mass of *A. marina*
278 leaves remained at the control and urban sites, respectively, while 17% and 11% were remaining
279 for *R. stylosa*, respectively.

280 Leaf litter mass loss is significantly more important after 56 and 72 days at the urban site
281 than at the control site ($p < 0.01$). Through the entire degradation experiment, the leaf litter mass
282 loss of *R. stylosa* is more important than that of *A. marina* ($p < 0.001$).

283 3.3. Carbon and nitrogen content, and respective stable isotope composition

284 The C and N content generally increased at both sites over time and for both species. At the
285 control site, the C content for both species and N content for *A. marina* increased the first 28 days,
286 and after 56 days. The N content of *R. stylosa* continuously increased during the 72 days of
287 degradation. At the urban site, the *R. stylosa* litterfall constantly increased in N and C contents. C
288 and N contents increased for *A. marina* until day 56 and 28, respectively, prior to a loss (Fig. 3A &
289 B). The C/N ratios for *A. marina* were stable the whole 72 days of degradation, while *R. stylosa* C/N
290 ratios continuously decreased with time (Fig. 3C).

291 The stable isotope ratios showed an initial enrichment for 7 days for $\delta^{15}\text{N}$ and 14 days for
292 $\delta^{13}\text{C}$ at both sites and for both species. Overall, after 72 days of degradation, the $\delta^{15}\text{N}$ of the litterfall
293 significantly decreased compared to the initial values ($p < 0.05$) except for *A. marina* at the urban
294 site where it increased (Fig. 3D). The $\delta^{13}\text{C}$ values significantly decreased for *R. stylosa* ($p < 0.001$)
295 while they were similar for *A. marina* at both sites after 72 days of degradation (Fig. 3E).

296 3.4. Molecular analyses

297 3.4.1. Neutral carbohydrates

298 After 72 days of litterfall degradation, total neutral carbohydrates content decreased from
299 452 to 380 mg g⁻¹ (-16%) and to 385 mg g⁻¹ (-15%) for *A. marina* at the control and urban sites,
300 respectively, and from 498 to 250 mg g⁻¹ (-50%) and to 388 mg g⁻¹ (-22%) for *R. stylosa* at the control
301 and urban sites, respectively (Fig. 4A). The most labile neutral carbohydrates after 72 days of
302 degradation were arabinose, rhamnose, galactose, and glucose with loss ranges of 23% to 80%,

303 2% to 76%, 31% to 50%, and 33% to 53%, respectively (Table 2). Xylose, ribose, fucose, and
304 mannose all increased their concentrations in the leaf litter after 72 days of degradation for most
305 stands reaching up to 152 mg g⁻¹, 1.2 mg g⁻¹, 10 mg g⁻¹, and 16 mg g⁻¹, respectively (Table 2).

306 For *A. marina*, the concentrations of pentoses (xylose + arabinose) and deoxy sugars
307 (rhamnose + fucose) in the senescent leaves (212 and 54 mg g⁻¹, respectively) are similar to the
308 concentrations in the 72 days-degraded leaves. For *R. stylosa*, the concentrations in the degraded
309 leaves are lower by 9% to 67% than the senescent leaves (235 mg g⁻¹ of pentoses and 66 mg g⁻¹ of
310 deoxy sugars). After 72 days of degradation, the concentrations of hexoses (mannose + galactose)
311 significantly decreased ($p < 0.001$) for the four stands by a range of 20% to 39% (Table 2).

312 3.4.2. Lignin-derived phenols

313 During the whole decay process, the sum of the lignin-derived phenols is significantly larger in
314 the *A. marina* litter than in *R. stylosa* litter ($p < 0.01$), whatever the site (Fig. 4B). This is also the case
315 for the four phenol groups (Table 3). After 7 days of degradation, the lignin content increased for *R.*
316 *stylosa* but decreased for *A. marina*. The total lignin content after 72 days of degradation also
317 increased for *R. stylosa* and decreased for *A. marina* compared to the lignin content in the senescent
318 leaves (Fig. 4B). This is only the case for the vanillyl phenols, while after 72 days of degradation the
319 syringyl phenols increased all between 71% and 231% (Table 3). In contrast to the total lignin
320 content, the ratios p-hydroxyl/vanillyl (P/V), syringyl/vanillyl (S/V), and cinnamyl/vanillyl (C/V)
321 increased after 7 and 72 days of degradation for *A. marina*, but decreased for *R. stylosa*, and
322 followed similar patterns with degradation at the two sites (Table 3). In the first days of degradation,
323 there was a decrease in the p-hydroxybenzoic acid/p-hydroxybenzaldehyde ratio ((Ad/Al)_p) at both
324 sites and a decrease in the vanillic acid/vanillin ratio ((Ad/Al)_v) at the urban site only. After 72 days
325 of degradation, the three acid/aldehyde ratios (Ad/Al) decreased at both sites and for both species
326 (Table 3).

327 4. Discussion

328 4.1. Variability of organic matter quality of the fresh and senescent leaves of *A. marina* and *R. stylosa*

329 Our investigation into the OM quality of leaves from *A. marina* and *R. stylosa* in urban and
330 control sites presented findings that connect with broader scientific contexts. Fresh leaves from the
331 urban site revealed high N and $\delta^{15}\text{N}$ values, aligning with expectations given the anthropogenic
332 influence in urban runoff. This is consistent with previous studies indicating elevated $\delta^{15}\text{N}$ values in
333 urban environments (Rumolo et al., 2011). Our study expanded upon this understanding by
334 demonstrating the species-specific nature of ^{15}N discrimination during N uptake, evident in the
335 greater $\delta^{15}\text{N}$ values observed in *A. marina* compared to *R. stylosa*, which was previously observed
336 for the same two genus in other studies (Herbon and Nordhaus, 2013; Regina Hershey et al., 2021).

337 The isotopic values, particularly $\delta^{13}\text{C}$ mean values of fresh leaves, provided valuable insights
338 into the adaptive responses of mangrove trees. Coastal plants, including mangrove trees, often
339 exhibit enhanced water-use efficiency under salinity stress. Our findings, showing lower $\delta^{13}\text{C}$ values
340 in the urban site and in *R. stylosa*, contributed to the understanding of these adaptive mechanisms
341 (Medina and Francisco, 1997). Plants have a decarboxylating enzyme that discriminates $^{13}\text{CO}_2$
342 when assimilating atmospheric CO_2 (O'Leary, 1981). The salinity of the soil and the position of the
343 mangrove stand in the mangrove forest may impact the mangrove species efficiency to assimilate
344 CO_2 or discriminate $^{13}\text{CO}_2$ (Farquhar et al., 1982; Lin and Sternberg, 1992). In New Caledonia, *A.*
345 *marina* grows on saltier soil than *R. stylosa* (26 and 37 psu, respectively at the control site) (Robin
346 et al., 2022) and therefore, has less $^{13}\text{CO}_2$ discrimination and higher $\delta^{13}\text{C}$ values. Moreover, the
347 upper soil at the urban site had a salinity of 0 psu for the first 10 cm (Robin et al., 2022), which may
348 explain the lower values measured.

349 As we explore the changes from fresh to senescent leaves at the control site we observed a
350 decline in N content, $\delta^{13}\text{C}$ values, and a notable rise in C/N ratios, especially in *R. stylosa* (up to
351 150%). These senescence-related shifts are connected with biochemical changes before leaf falling,
352 linking to established knowledge in mangrove ecology (Garten and Taylor, 1992; Rao et al., 1994;
353 Nordhaus et al., 2011). Prior to leaf falling, the mangrove trees can relocate the N from the leaves
354 to other tissues of the plant to limit N loss explaining the decline in N content. A general decrease
355 in C content during senescing can be expected due to the loss of photosynthetic activity, but the

356 relative increase in carbohydrates as a signal for senescing may induce the increase in C content
357 in some mangrove species as observed in this study for *A. marina* (Wingler et al., 2006).

358 Examining lignin-derived phenols and neutral carbohydrates content, our study
359 contextualized findings within the existing literature on mangrove leaf composition and degradation
360 dynamics (Marchand et al., 2005; Lallier-Vergès et al., 2008; Bala Krishna Prasad and Ramanathan,
361 2009). Higher lignin and neutral carbohydrate content in the fresh leaves of *A. marina* compared to
362 *R. stylosa* agree with previous studies (Dittmar and Lara, 2001). However, the higher proportions of
363 hexoses and deoxy sugars in the leaves of *R. stylosa*, especially fucose and galactose, underscored
364 the need for a comprehensive insight of initial leaf compositions. The production or loss of neutral
365 carbohydrates and lignin-derived phenols during senescence contribute to our knowledge of the
366 intricate senescence process in mangrove leaves (Benner et al., 1990b; Opsahl and Benner, 1995,
367 1999).

368 4.2. Factors influencing leaf litter degradation between mangrove species

369 The leaf litter of *R. stylosa* was degraded more rapidly on the soil surface than the leaf litter
370 of *A. marina* at both sites; a trend opposite to conventional findings where species with higher initial
371 N content, and consequently lower C/N ratios, typically undergo faster degradation (Mfilinge et al.,
372 2002; Yang et al., 2018; Vinh et al., 2020). Notably, *A. marina*, initially possessed five times more N
373 in its senescent leaves than *R. stylosa*. The higher N content in plant material is known to attract
374 leaf-degrading organisms due to the nutritional value of N compounds (Wafar et al., 1997; Nordhaus
375 et al., 2011). We propose that the species' position within the mangrove forest is a crucial
376 determinant of litterfall decay rates. Generally, species the most submerged experience faster
377 degradation rates (Twilley et al., 1986; Steinke and Ward, 1987; Silva et al., 1998; Bosire et al.,
378 2005). The physical impact of tides facilitates plant material breakdown (Vinh et al., 2020), and
379 submersion by seawater or runoff promotes microbial development and leaching of soluble
380 compounds from litterfall (Robertson, 1988; Tam et al., 1990; Ashton et al., 1999). In the mangrove
381 forests of New Caledonia, *R. stylosa* is found on the seaward side of the mangrove forest, while *A.*
382 *marina* is present at higher elevations with higher soil salinity (Deborde et al., 2015). However, in
383 the urban mangrove forest, continuous runoff from urban rainwater altered soil physico-chemical

384 parameters, particularly pH and salinity (0 psu in the upper 10 cm) (Robin et al., 2022).
385 Consequently, *R. stylosa* trees are established on the landside and at the entrance of the runoff in
386 the mangrove, providing a subtidal context unique to the urban site. Therefore, submersion of the
387 *R. stylosa* stand by seawater at the control site and submersion by the urban runoff at the urban site
388 may contribute to enhanced litterfall degradation for *R. stylosa*, by favoring leaching and microbial
389 development on the leaf litter compared to *A. marina* (Robertson, 1988; Molnar et al., 2014).

390 Despite the unexpected degradation trend observed in the leaf litter of *R. stylosa* compared
391 to *A. marina*, further insights into the isotopic dynamics shed light on unique microbial processes
392 driving the decomposition of litterfall of *R. stylosa*. The statistically significant decrease in $\delta^{15}\text{N}$
393 values with leaf litter degradation was only observed for *R. stylosa*. Bacteria dominating
394 decomposition processes have low $\delta^{15}\text{N}$ values as bacteria favour the immobilization of ^{15}N -depleted
395 nitrate (Bragazza et al., 2010; Nordhaus et al., 2011). The results therefore suggest that microbial
396 development beneath the *R. stylosa* stands at both sites was enhanced, possibly due to their
397 position compared to the water sources, and can therefore ultimately lead to the lowering of
398 degrading leaf litter $\delta^{15}\text{N}$ values (Leopold et al., 2013; Molnar et al., 2014).

399 Leaf litter composition changes during degradation, specifically the N content, C/N ratio, and
400 isotopic values ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$), were investigated. Degradation of the leaf litter of *R. stylosa* resulted
401 in a decrease in the C/N ratio not observed for the leaf litter of *A. marina*. The decreasing ratio for
402 *R. stylosa* may be associated with the immobilization of N by bacteria as observed in a variety of
403 tropical environments (Gotto and Taylor, 1976; Rice and Tenore, 1981; Twilley et al., 1986; Benner
404 et al., 1990a). As suggested in the previous paragraph, the decreasing ratio may only be observed
405 for *R. stylosa* as microbial development beneath the *R. stylosa* stands at both sites was possibly
406 enhanced. The decrease in the C/N ratio may also be partly linked to the degradation of hydrolysable
407 components, particularly neutral sugars, rather than refractory N-rich components like lignin (Chale,
408 1993; Yang et al., 2018). Indeed, the neutral carbohydrate content of the leaf litter of *R. stylosa*
409 decreased during degradation by 22% and 50% at the urban and control sites, respectively, while
410 the lignin content increased after 7 and 72 days of degradation. Therefore, bulk leaf decayed more
411 rapidly than lignin, as reported in the literature for anaerobic and sulphate reducing environments

412 (Benner et al., 1984, 1991; Dittmar and Lara, 2001). Conversely, the lignin content decreased in the
413 leaf litter of *A. marina*. The upper soil beneath *A. marina* was less reducing than *R. stylosa* (Robin
414 et al., 2022), which can favour lignin degradation (Benner et al., 1991; Opsahl and Benner, 1995).
415 *A. marina* also had a higher percentage of lignin content in its senescent leaves than *R. stylosa*,
416 suggesting that the initial molecular composition of the leaves influenced the lability of the molecules
417 during degradation. Higher lignin content may enhance lignin degradation by favouring microbial
418 access to lignin polymers. This phenomenon, where the concentrations of the molecules in
419 senescent leaves influence their decays, aligns with findings in terrestrial plants (Talbot et al., 2012).

420 4.3. Urban runoff effects on leaf litter degradation

421 Urban runoff and associated modified conditions at the urban site seemed to affect leaf litter
422 degradation rates as the mass loss was faster in the urban mangrove forest compared to the control
423 mangrove forest. This discrepancy in mass loss can be attributed to the continuous flooding of the
424 forest floor by urban rainwater runoff, which could enhance the early leaching process of the leaf
425 litter. While previous studies have reported faster mass loss in subtidal stands compared to intertidal
426 stands (Robertson et al., 1992), our findings reveal that, after 7 days of degradation, there is no
427 significant difference in leaf litter mass loss between the urban and control sites. However, it is after
428 56 days of leaf litter degradation that statistical differences in mass loss emerge between the urban
429 and control sites. This suggests that the second phase of degradation, involving the breakdown of
430 litter by microorganisms, becomes more prominent at the urban site. Therefore, we suggest that
431 urban runoff likely favors microbial activity on the soil surface, leading to the observed divergence
432 in mass loss between the two sites during the later stages of leaf litter degradation. The runoff gives
433 a more stable, nutritious, and humid environment for microbial development (Robertson, 1988;
434 Leopold et al., 2013; Molnar et al., 2014). Mineralization of OC in coastal wetland soils is significantly
435 influenced by temperature and soil water content, which regulate microbial metabolism (Marchand
436 et al., 2022a). Additionally, it is impacted by the rate of electron acceptor renewal, varying with
437 immersion. As the duration of immersion increases, the mineralization process may be enhanced or
438 reduced, depending on soil permeability (Marchand et al., 2022b).

439 The influence of urban runoff on leaf litter degradation was also evident in the larger increase
440 in N content during degradation at the urban site than the control site. Urban rainwater runoff may
441 have acted as an additional source of N to the forest floor and litter (Tam et al., 1998). Nitrogen can
442 also be produced by microorganisms during leaf litter degradation (Tam et al., 1990; Mfilinge et al.,
443 2002). The higher degradation rates associated with the higher enrichment of fucose and ribose at
444 the urban site, characteristic of microbial production (Cowie and Hedges, 1984), indicated stronger
445 microbial activity on the soil surface at the urban site.

446 The modified soil conditions created by the submerging urban runoff may also have
447 influenced molecular degradation processes of the leaf litter. After 72 days of degradation rhamnose
448 and glucose loss was more important at the control site than at the urban site. With a lower soil
449 surface salinity (Robin et al., 2022), the urban mangrove forest was a less suitable environment for
450 the secretion of some enzymes such as cellulases, which enhance cellulose and hemicellulose
451 hydrolysis (Arfi et al., 2013; Ivaldi et al., 2021). Also, the more anoxic environment of the urban
452 forest's upper soil (Robin et al., 2022) limited the hydrolysis of neutral carbohydrates including
453 rhamnose and glucose. The more reducing conditions may also have favoured lignin conservation
454 over bulk OC during the early phase of degradation (Benner et al., 1984, 1991), explaining the higher
455 relative lignin enrichment in the leaf litter of *R. stylosa* at the urban site than at the control site after
456 7 days of degradation.

457 *4.4. Leaf litter molecular composition during degradation*

458 The molecular-level changes of OM during leaf litter degradation revealed species-specific
459 and site-dependent influences on the stability of some molecules. In this study, we observed that
460 the most abundant sugars, that is glucose, arabinose, galactose, and rhamnose, were the most
461 labile, and the least abundant, that is sugars fucose, ribose, and mannose, were the most refractory
462 ones. Trends like this one were observed in neutral carbohydrates and lignin-derived phenols during
463 degradation, suggesting a potential end for compositional uniformity (Opsahl and Benner, 1999).
464 The hypothesis of compositional uniformity suggests that the most abundant sugars would be more
465 labile than the least abundant in order to end up with a uniform composition of neutral carbohydrates
466 in the most degraded leaves (Opsahl and Benner, 1999). Except for xylose in this study, results

467 seem to agree with this hypothesis. We suggest that xylose may be associated to more refractory
468 components of the litter such as lignin, which enhanced protection from enzymatic degradation
469 (Opsahl and Benner, 1999), as suggested by the correlation between xylose and p-hydroxyl phenols
470 ($r = 0.61$) and cinnamyl phenols ($r = 0.50$) (Table S7).

471 The order of lability of the 11 lignin-derived phenols differed between mangrove species and
472 sites but observations agreed with previous work. In the literature, syringyl phenols were classified
473 as the most stable phenols and ferulic acid the least (Benner et al., 1990b). In the present study,
474 ferulic acid content per g of TOC decreased in the leaf litter at the four stands after 72 days of
475 degradation (Table S8), while total syringyl phenols increased. The hypothesis of compositional
476 uniformity may also work here as for both species, the most abundant phenol group in the senescent
477 leaves was also the most labile during degradation; that is vanillyl phenols for *A. marina* and
478 cinnamyl phenols for *R. stylosa*.

479 4.5. Limitations of the experiment and models

480 The litterbag experiment was conducted using senescent leaves from the control site only
481 for both species. This design of experiment aimed to evaluate the changes in leaf litter mass and
482 composition over 72 days of degradation with a focus on the influence of urban runoff on these
483 changes. The choice of having the same senescent leaves at both sites enabled to limit the factors
484 potentially affecting these changes other than runoff such as initial senescent leaves composition.
485 The influence of initial composition was evaluated between species but not between sites. This
486 experiment design was therefore not a limitation regarding the general processes investigated in
487 this study. However, it is not possible from this experiment to quantify the elements or molecules
488 released, stored, or gained by the leaf litter at the urban site. Furthermore, the experiment ended
489 after 72 days of degradation as the mass remaining was the lower limit to conduct all the analyses.
490 A longer experiment could have given further information relative to the refractory degradation stage.

491 In this study, single exponential models were computed on the mass loss of the leaf litter
492 with degradation. Single exponential models were used to calculate the half-lives of the litter and
493 are frequently used in similar works using the litterbag experiment (Flores-Verdugo et al., 1987;

494 Ashton et al., 1999; Bosire et al., 2005; Nordhaus et al., 2017; Vinh et al., 2020). However, double
495 exponential models seemed more suitable for the degradation rate in this study as shown by the
496 regression coefficients (R^2) of the exponential regression lines (Figure S1), as previously suggested
497 in the literature (Wider and Lang, 1982). Two phases of degradation stood out in the double
498 exponential model with a first stage from 0 to 14 days for *R. stylosa* or to 28 days for *A. marina*, and
499 a second to 72 days (Figure S1). The double exponential model indicated that the mass loss of the
500 leaf litter was faster for the four stands during the first days of degradation as soluble compounds
501 were leached. Still, the use of the single exponential model is preferential as comparison with
502 literature is possible and the differences between species and sites were noticeable as much as with
503 the double exponential model.

504 **5. Summary and conclusions**

505 In this study, changes in mangrove leaf litter OM quality during degradation was analysed in
506 relation to mangrove species (*A. marina* and *R. stylosa*) and to mangrove forests (urban and control)
507 with a focus on the potential influence of urban runoff. Considering mangrove species, we were
508 surprised to observe that despite lower initial N content, degradation of *R. stylosa* leaf litter was
509 faster than the one of *A. marina* at both sites. We suggest that species position in the intertidal zone
510 is crucial and that length of immersion, either tidal or runoff, are key factors controlling degradation
511 rates. The immersion not only influences the leaching but also enhances the microbial development,
512 as evidenced by $\delta^{15}\text{N}$ values variations beneath both *R. stylosa* stands. Regarding mangrove
513 forests, the urban runoff submerging the mangrove soil amplified leaf litter degradation as shown by
514 the lower half-lives (about ten days) for both species compared to the control site. We suggest that
515 the runoff enhanced both leaching and microbial activity. Further experiments in mesocosms with
516 controlled parameters should be undertaken to better understand the decay processes generated
517 by the runoff. The physico-chemical conditions (more reducing, lower salinity, ...) at the urban site
518 not only influenced mass loss but also shaped the molecular composition of the OM during leaf litter
519 degradation. The lower soil surface salinity may have limited enzyme secretion and hydrolysis of
520 neutral carbohydrates. While urban runoff and site conditions impact leaf litter degradation, this
521 study highlights that the initial molecular composition of the litter also plays a crucial role in

522 controlling its molecular evolution during degradation. The observed leaf litter composition in neutral
523 carbohydrates and lignin-derived phenols during degradation underscored the intricate interplay
524 between environmental factors and OM stability in mangrove ecosystems.

525 These findings have broader implications for understanding the global impacts of urban
526 runoff on mangrove ecosystems, including potential reductions in carbon sequestration capacity and
527 threats to ecosystem services such as habitat provision, fisheries support, and coastal erosion
528 mitigation.

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535 **Data Availability**

536 Data are available through Mendeley Data, V2 at <https://doi:10.17632/hh48m5g37v.2>.

537 **Appendix A. Supplementary Material**

538 Supplementary data referred in the text are available as supplementary material. Information on the
539 instrument methods and quality control is available (GC-MS and UPLC-MS).

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733 species. *Estuar. Coast. Shelf Sci.* **214**, 141–148.

734 Tables

735 Table 1. Fresh and senescent leaves characteristics (mean value \pm SD).

Species	<i>A. marina</i>			<i>R. stylosa</i>		
	Control		Urban	Control		Urban
Site						
Leaf	Fresh	Senescent	Fresh	Fresh	Senescent	Fresh
C (g kg ⁻¹)	404 \pm 22	441 \pm 3	416 \pm 23	407 \pm 23	374 \pm 3	422 \pm 24
N (g kg ⁻¹)	19 \pm 2	11 \pm 1	24 \pm 2	7.6 \pm 1.5	2.3 \pm 0.4	15 \pm 2
C/N	21 \pm 4	39 \pm 2	17 \pm 3	54 \pm 14	166 \pm 30	28 \pm 5

$\delta^{13}\text{C}$ (‰vsVPDB)	-25.8±0.3	-27.7±0.0	-27.7±0.7	-27.5±0.1	-28.3±0.3	-30.7±1.0
$\delta^{15}\text{N}$ (‰vsAir)	1.1±0.6	0.9±0.7	5.8±1.7	-1.3±0.5	-0.2±0.2	2.6±0.1
$\Sigma 11$ phenols (mg g ⁻¹)	4.1±1.2	43±11	5.0±1.3	1.3±0.3	6.6±2.2	1.4±0.5
p-hydroxyl (mg g ⁻¹)	0.62±0.10	0.27±0.03	0.56±0.11	0.21±0.02	0.15±0.00	0.14±0.04
Vanillyl (mg g ⁻¹)	2.0±0.6	33±10	2.0±0.4	0.56±0.27	0.90±0.17	0.58±0.16
Syringyl (mg g ⁻¹)	0.33±0.07	0.22±0.00	0.26±0.04	0.22±0.02	0.14±0.06	0.20±0.04
Cinnamyl (mg g ⁻¹)	1.1±0.1	9.2±1.1	2.1±0.9	0.34±0.06	5.4±2.3	0.48±0.24
P/V	0.31	0.01	0.28	0.38	0.17	0.24
S/V	0.17	0.01	0.13	0.39	0.16	0.34
P/(V+S)	0.27	0.01	0.25	0.27	0.14	0.18
(Ad/Al) _p	0.2	2.57	0.2	0.3	4.58	0.3
(Ad/Al) _v	0.8	10.98	3.3	1.1	20.88	2.1
(Ad/Al) _s	1.2	2.99	1	3.2	1.36	1.7
Neutral carbohydrates (mg g ⁻¹)	384	452±52	322	228	499±14	304
Xylose (mg g ⁻¹)	128	122±23	61	49	104±7	66
Arabinose (mg g ⁻¹)	97	90±14	62	62	131±6	102
Ribose (mg g ⁻¹)	1.9	1.0±0.1	1.6	0.48	0.76±0.07	1.4
Rhamnose (mg g ⁻¹)	41	52±6	29	27	58±1	37
Fucose (mg g ⁻¹)	1.4	1.9±0.5	0.71	3.8	8.0±0.2	4.5
Mannose (mg g ⁻¹)	10	11±2	5.1	4.1	11±1	5.6
Galactose (mg g ⁻¹)	38	42±6	28	34	69±3	61
Glucose (mg g ⁻¹)	66	133±1	135	48	118±3	27

736 11 phenols = p-hydroxybenzoic acid, p-hydroxybenzaldehyde, p-hydroxyacetophenone, vanillic acid, vanillin,
737 acetovanillone, syringic acid, syringaldehyde, acetosyringone, coumaric acid, and ferulic acid; p-hydroxyl = p-
738 hydroxybenzoic acid, p-hydroxybenzaldehyde, and p-hydroxyacetophenone; vanillyl = vanillic acid, vanillin, and
739 acetovanillone; syringyl = syringic acid, syringaldehyde, and acetosyringone; cinnamyl = coumaric acid and ferulic acid;
740 P/V = ratio of p-hydroxyl over vanillyl; S/V = ratio of syringyl over vanillyl; P/(V+S) = ratio of p-hydroxyl over the sum of
741 vanillyl and syringyl; (Ad/Al) = acid/aldehyde.

742 Table 2. Mean neutral carbohydrate concentrations (±SD) in mg g⁻¹ in leaf litter with degradation at
743 the control and urban sites for both *A. marina* and *R. stylosa*.

Control <i>A. marina</i>	senescent	7 days	14 days	28 days	56 days	72 days
Xylose	122±23	127±11	97±2	122±1	167±1	152±0
Arabinose	90±14	85±4	57±5	55±1	76±3	70±6
Ribose	1.0±0.1	1.0±0.3	0.71±0.01	0.52±0.20	0.94±0.02	0.83±0.22
Rhamnose	52±6	52±5	37±2	36±1	47±1	45±5
Fucose	1.9±0.5	2.0±0.5	1.7±0.0	1.8±0.1	2.3±0.5	2.5±0.9
Mannose	11±2	18±3	11±0	9.3±2.2	14±5	15±8
Galactose	42±6	41±3	21±2	16±2	24±5	25±9
Glucose	133±1	103±5	70±0	58±0	74±3	71±9
Control <i>R. stylosa</i>	senescent	7 days	14 days	28 days	56 days	72 days
Xylose	104±7	73±1	83±17	85±12	86±4	78±5
Arabinose	131±6	76±2	73±10	63±4	50±3	46±2
Ribose	0.76±0.07	0.39±0.07	0.46±0.05	0.48±0.06	0.40±0.02	0.41±0.05
Rhamnose	58±0	33±1	33±6	25±4	15±2	14±1
Fucose	8.0±0.2	6.0±0.7	6.3±1.5	6.9±0.2	7.5±0.3	8.0±0.9
Mannose	11±0	8.6±0.4	8.3±3.5	11±0	11±2	14±4
Galactose	69±3	48±2	47±11	42±2	34±5	34±6
Glucose	117±3	61±11	62±22	56±2	50±10	55±9
Urban <i>A. marina</i>	senescent	7 days	14 days	28 days	56 days	72 days
Xylose	122±23	150±6	108±10	134±4	157±14	145±22
Arabinose	90±14	100±2	59±4	67±5	76±9	72±9
Ribose	1.0±0.1	1.3±0.3	1.0±0.0	1.5±0.0	1.2±0.1	1.2±0.1

Rhamnose	52±6	64±5	38±1	52±0	50±14	50±0
Fucose	1.9±0.5	2.8±0.8	1.5±0.0	2.6±0.5	2.2±0.1	2.1±0.0
Mannose	11±2	20±5	11±2	21±3	16±1	13±1
Galactose	42±6	47±9	21±1	30±1	26±2	22±1
Glucose	133±1	118±2	67±6	87±4	79±21	80±3
Urban <i>R. stylosa</i>	senescent	7 days	14 days	28 days	56 days	72 days
Xylose	104±7	113±11	96±3	127±4	111±10	135±6
Arabinose	131±6	115±9	83±10	71±17	58±13	78±10
Ribose	0.76±0.07	0.75±0.03	0.70±0.07	1.1±0.0	0.96±0.21	1.2±0.4
Rhamnose	58±0	51±8	36±0	21±15	19±2	21±4
Fucose	8.0±0.2	9.3±0.3	10±0	14±3	7.5±1.0	10±1
Mannose	11±0	13±1	12±1	33±29	15±3	16±3
Galactose	69±3	72±8	55±0	66±14	38±4	47±3
Glucose	117±3	91±10	76±1	30±40	64±8	79±9

744

745 Table 3. Lignin-derived phenols parameters in leaf litter with degradation at the control and urban
746 sites for both *A. marina* and *R. stylosa* (mean values ± SD).

Control <i>A. marina</i>	senescent	7 days	14 days	28 days	56 days	72 days
p-hydroxyl (mg g ⁻¹)	0.27±0.03	0.15±0.03	0.20±0.02	0.22±0.04	0.26±0.07	0.26±0.04
Vanillyl (mg g ⁻¹)	33±10	18±14	15±2	10 ±2	4.4±0.9	14±3
Syringyl (mg g ⁻¹)	0.22±0.001	0.14±0.05	0.31±0.21	0.42±0.10	0.47±0.26	0.50±0.05
Cinnamyl (mg g ⁻¹)	9.2±1.1	6.8±0.7	6.8±0.5	11±2	12±2	12±2
P/(V+S)	0.029	0.022	0.028	0.020	0.021	0.022
P/V	0.008	0.008	0.013	0.021	0.058	0.019
S/V	0.007	0.008	0.021	0.041	0.106	0.037
C/V	0.27	0.38	0.45	1.00	2.68	0.86
(Ad/Al) _p	2.57	2.01	1.50	0.54	0.49	0.52
(Ad/Al) _v	10.98	11.62	5.57	0.89	0.70	1.10
(Ad/Al) _s	2.99	2.53	0.52	0.39	0.45	0.47
Control <i>R. stylosa</i>	senescent	7 days	14 days	28 days	56 days	72 days
p-hydroxyl (mg g ⁻¹)	0.15±0.001	0.12±0.02	0.13±0.01	0.11±0.004	0.14±0.01	0.13±0.004
Vanillyl (mg g ⁻¹)	0.90±0.17	5.7±4.1	2.6±1.5	2.2±0.9	3.3±1.2	3.4±0.4
Syringyl (mg g ⁻¹)	0.14±0.06	0.12±0.01	0.14±0.03	0.16±0.04	0.20±0.03	0.24±0.02
Cinnamyl (mg g ⁻¹)	5.4±2.3	4.2±0.5	3.8±0.3	3.0±0.3	4.7±0.9	4.9±0.7
P/(V+S)	0.026	0.027	0.034	0.033	0.029	0.025
P/V	0.160	0.021	0.051	0.048	0.042	0.039
S/V	0.152	0.021	0.054	0.073	0.061	0.073
C/V	5.99	0.75	1.45	1.36	1.42	1.45
(Ad/Al) _p	4.58	3.71	2.32	2.38	1.82	1.78
(Ad/Al) _v	20.88	19.53	34.32	26.32	19.50	13.59
(Ad/Al) _s	1.36	1.08	1.12	1.38	1.40	1.10
Urban <i>A. marina</i>	senescent	7 days	14 days	28 days	56 days	72 days
p-hydroxyl (mg g ⁻¹)	0.27±0.03	0.23±0.005	0.23±0.08	0.21±0.02	0.28±0.03	0.30±0.16
Vanillyl (mg g ⁻¹)	33±10	7.5±6.0	15±9	11±5	16±6	23±16
Syringyl (mg g ⁻¹)	0.22±0.001	0.39±0.01	0.68±0.22	0.53±0.19	0.61±0.15	0.73±0.35

Cinnamyl (mg g ⁻¹)	9.2±1.1	9.1±4.1	10±4	9.1±2.1	14±1	9.7±4.4
P/(V+S)	0.029	0.024	0.021	0.021	0.020	0.029
P/V	0.008	0.030	0.015	0.019	0.017	0.013
S/V	0.007	0.051	0.045	0.049	0.038	0.031
C/V	0.27	1.21	0.68	0.85	0.85	0.42
(Ad/Al) _p	2.57	0.56	1.00	0.66	0.86	0.61
(Ad/Al) _v	10.98	2.58	3.21	1.83	4.09	2.46
(Ad/Al) _s	2.99	0.48	0.43	0.62	0.64	0.60
Urban <i>R. stylosa</i>	senescent	7 days	14 days	28 days	56 days	72 days
p-hydroxyl (mg g ⁻¹)	0.15±0.001	0.20±0.05	0.12±0.02	0.17±0.03	0.23±0.07	0.18±0.07
Vanillyl (mg g ⁻¹)	0.90±0.17	11±15	3.5±3.2	2.6±0.9	6.8±3.5	2.6±1.2
Syringyl (mg g ⁻¹)	0.14±0.06	0.17±0.02	0.12±0.09	0.22±0.17	0.11±0.03	0.23±0.08
Cinnamyl (mg g ⁻¹)	5.4±2.3	3.8±0.6	2.4±0.3	3.5±1.6	4.6±1.6	6.0±1.7
P/(V+S)	0.026	0.051	0.049	0.045	0.050	0.029
P/V	0.160	0.018	0.035	0.064	0.034	0.071
S/V	0.152	0.015	0.036	0.085	0.017	0.091
C/V	5.99	0.34	0.69	1.34	0.67	2.34
(Ad/Al) _p	4.58	1.86	1.90	1.37	1.60	1.49
(Ad/Al) _v	20.88	6.25	10.69	4.63	2.55	6.25
(Ad/Al) _s	1.36	2.31	1.36	1.47	1.09	0.56

747 P = p-hydroxyl, V = Vanillyl, S = Syringyl, C = cinnamyl, Ad = acid, Al = aldehyde.

748 **Figure captions**

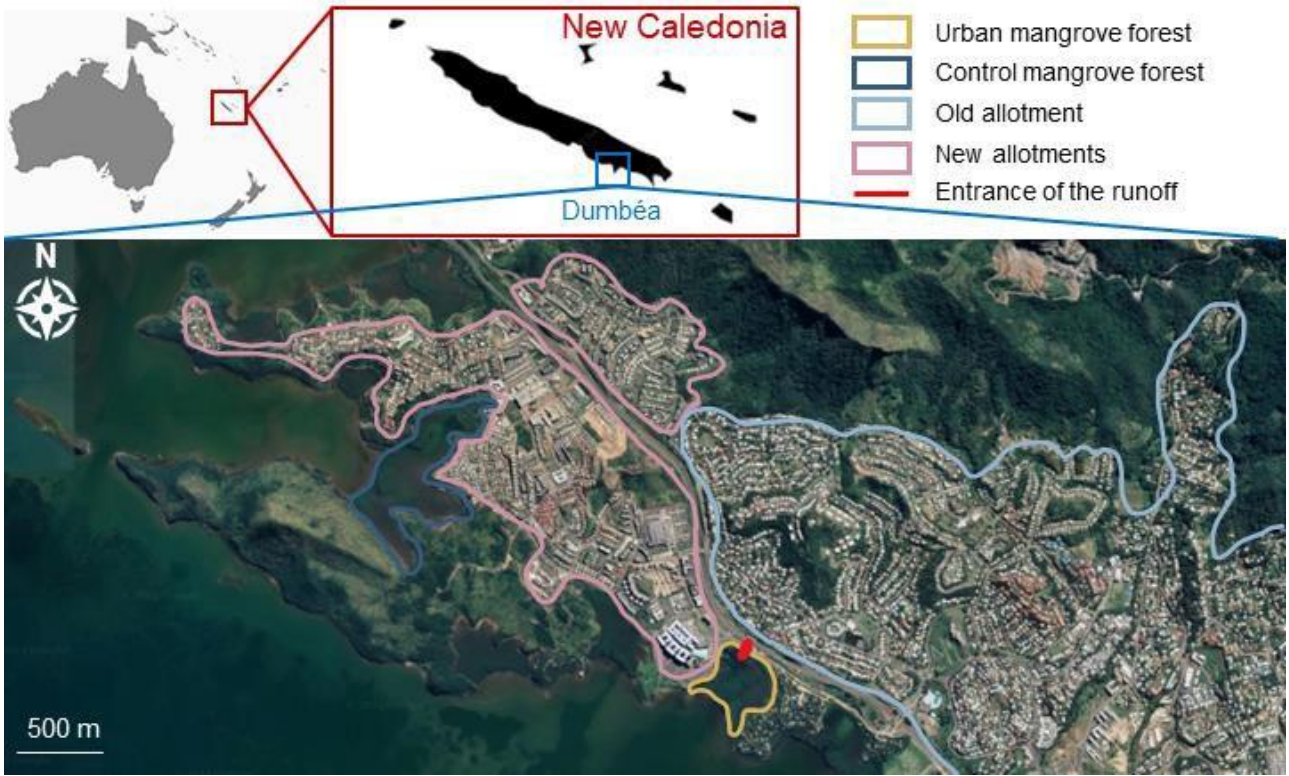
749 Figure 1. Control mangrove forest and urban mangrove forest in Dumbea city in New Caledonia with
750 delimitation of the new allotments and the old allotment and the entrance of the urban rainwater
751 runoff.

752 Figure 2. Decay rates of leaf litter degradation represented by the mean mass remaining of the leaf
753 litter in the litterbags during degradation at the control site for (A) *A. marina* and (B) *R. stylosa* and
754 at the urban site for (C) *A. marina* and (D) *R. stylosa*. The error bars represent the standard
755 deviations.

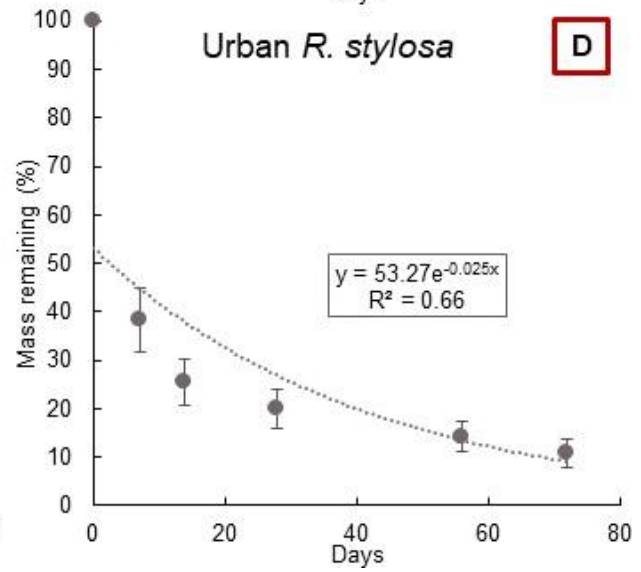
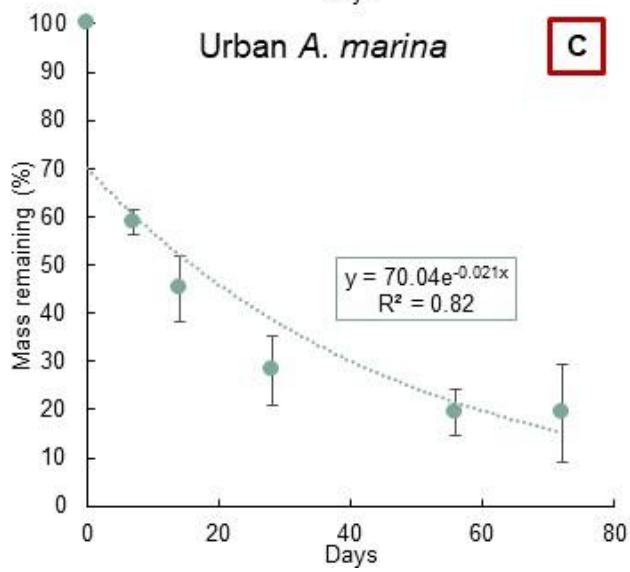
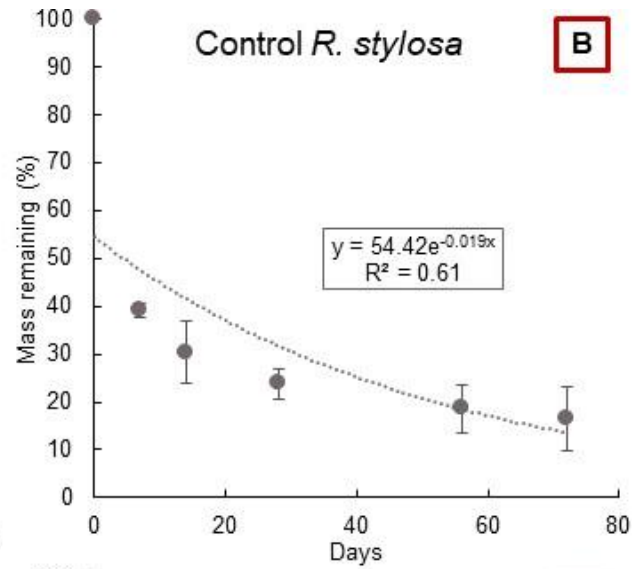
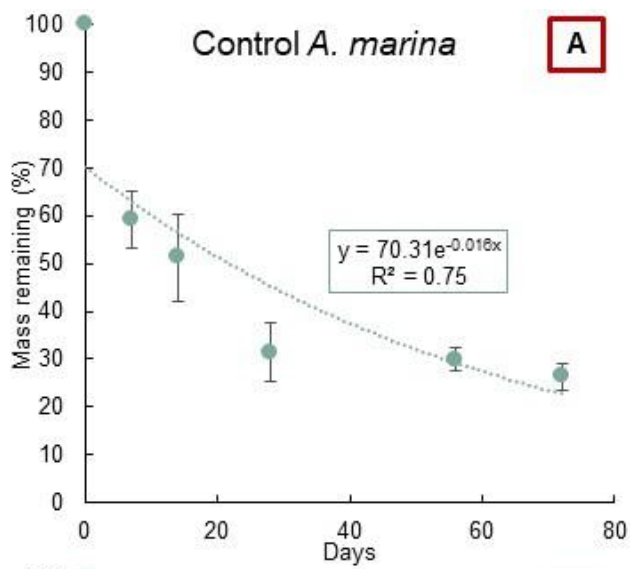
756 Figure 3. Mean values (±SD) of (A) C in g kg⁻¹, (B) N in g kg⁻¹, (C) C/N ratio, (D) δ¹⁵N (‰vsAir) and
757 (E) δ¹³C (‰vsVPDB) during leaf litter degradation for *A. marina* and *R. stylosa* at the control and
758 urban sites.

759 Figure 4. (A) Mean (±SD) total neutral carbohydrate content in mg g⁻¹ in leaf litter with degradation
760 at the control and urban sites and for *A. marina* and *R. stylosa* and (B) mean (±SD) sum of the 11
761 lignin-derived phenols in mg g⁻¹.

762 **Figures**

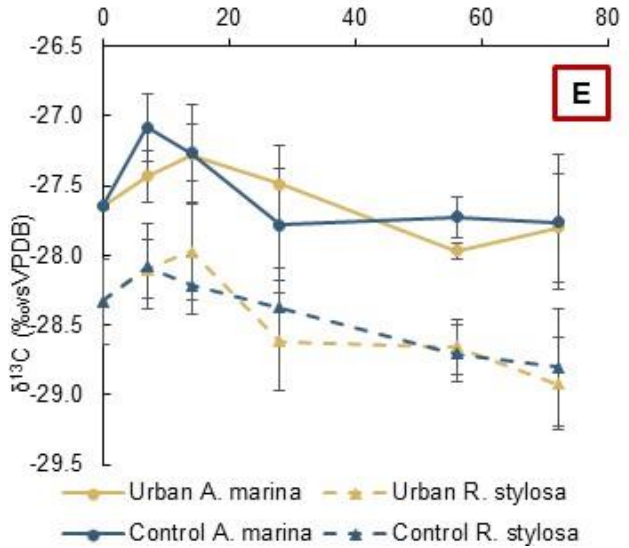
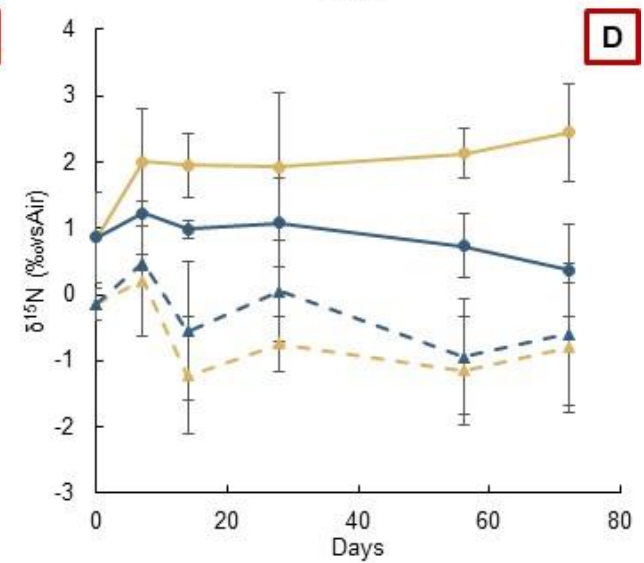
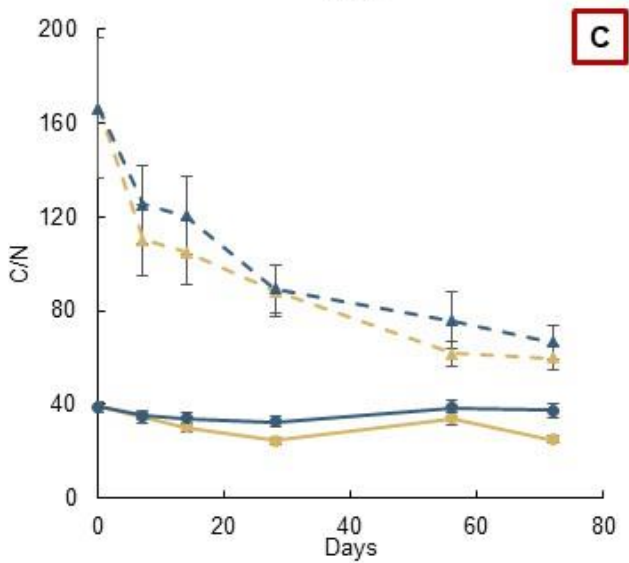
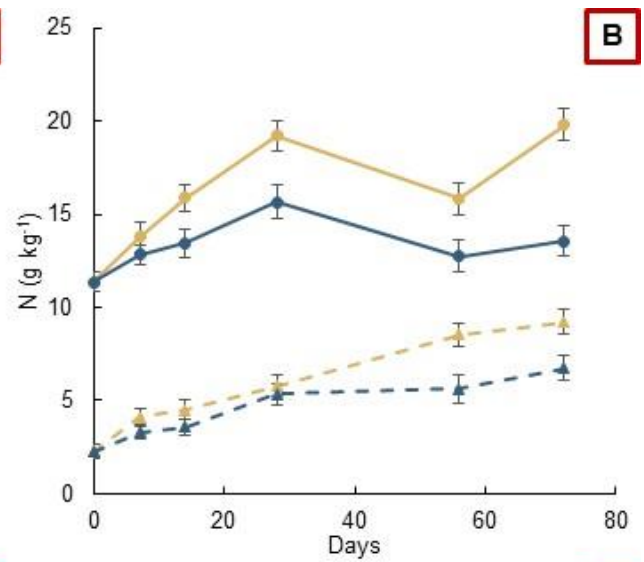
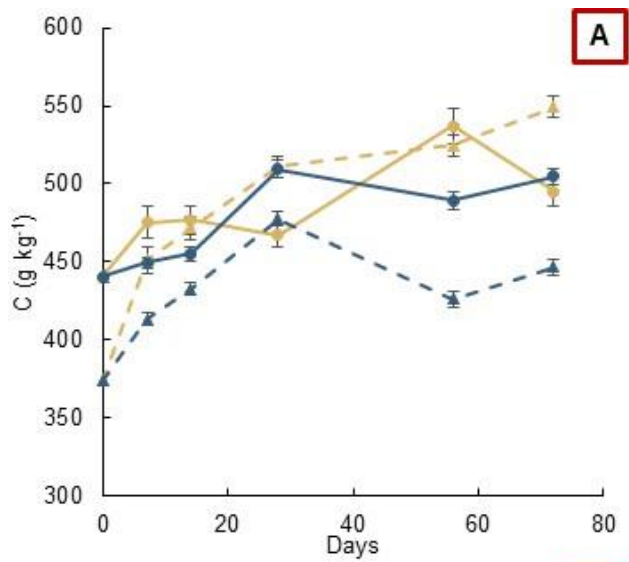


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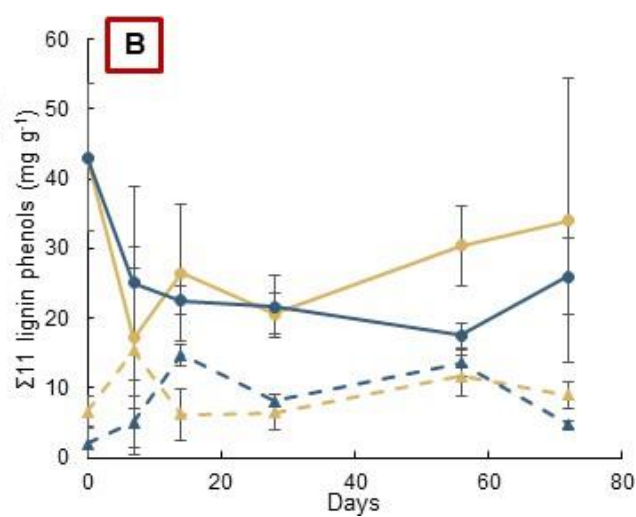
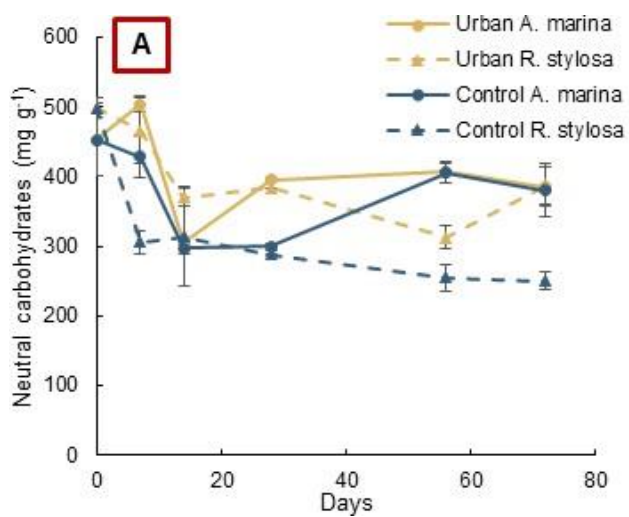


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—●— Urban *A. marina* - -■- - Urban *R. stylosa*
—●— Control *A. marina* - -■- - Control *R. stylosa*



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