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Published paper

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1 **Are calcareous soils in uplands less prone to damage from road salting than**
2 **acidic soils?**

3
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6

7 Previous studies of upland roadside soils in Cumbria, that would normally be naturally
8 acidic, have highlighted that (a) runoff from roads subjected to long-term road salting can
9 dramatically raise soil pH down slope in upland areas; (b) the soil pH increase dramatically
10 changes N cycling in soils down slope, increasing mineralization of organic matter,
11 ammonification, ammonium leaching down slope and nitrification and nitrate leaching; (c)
12 the increase in nitrification substantially increases nitrate leaching to down-slope rivers, and
13 this is readily detectable in field studies; and (d) loss of soil organic matter over decades of
14 salting is so great that organic matter is no longer substantially solubilized by high salt
15 concentrations found in soil solution below road drains. This paper tests and supports the
16 hypothesis that such effects are minimal for more calcareous soil ecosystems. It examines
17 the soil and soil solution chemistry on another Cumbrian upland highway, the A686 near
18 Leadgate, Alston. Sodium % of soil CEC values for soil transects affected by spray
19 containing road salt are similar at both the A6 and A686 sites. However, spatial trends in
20 calcium, magnesium, ammonium, and nitrate concentrations as well as pH differ, as a direct
21 result of the higher weathering rate of parent material and possibly also the presence of
22 limestone walls above both spray-affected and control transects at the A686 site.
23

24 *Keywords:* Road salt; sodium; chloride; base cations; weathering rate; N cycling

25 **1. Introduction**

26 De-icing agents are widely applied to roads around the world under sub-zero temperature or
27 snowy conditions to prevent the formation (or reformation) of ice and thus maintain safe
28 driving conditions. Materials used include rock salt (sodium chloride), calcium chloride,

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29 calcium magnesium acetate (CMA), magnesium chloride, potassium chloride, sodium formate
30 and urea [1-4]. Sodium chloride tends to be the preferred agent in the UK, applied in solution
31 or as a solid, depending upon the conditions. It may be mixed with grit, and sometimes
32 contains an anti-caking agent [5,6]. Much of the applied material ends up being transferred to
33 roadside soils, especially down-slope of the highway.

34 Road salt is used on 64.5% of roads in the County of Cumbria in North West England.
35 Thus for 2262 km of road in the county, episodically the roadside soils are exposed to
36 incoming road salt (i.e. impure NaCl) via spray, runoff and drainage. Green and Cresser [7,8]
37 and Green *et al.* [9-11] assessed the impacts of road salt on a soil that would naturally be very
38 acidic adjacent to a highway, on its soil solution, and on water in an adjacent, down-slope
39 river. The study site at the A6, Selside, Cumbria experiences directly piped runoff onto the soil
40 surface and/or spray. They reported the following main findings:

- 41 1. Runoff from an upland road subjected to long-term road salting dramatically
42 raised soil pH down slope [7].
- 43 2. The pH increase markedly changed N cycling in soils down slope, increasing
44 mineralization of organic matter, ammonification and nitrification and nitrate
45 leaching [8, 10]. Further modification was caused by displacement of
46 ammonium by sodium ions and mobile anion effects on soil H and cation
47 mobility's.
- 48 3. The increase in nitrification substantially increased nitrate leaching to a down-
49 slope river, and this was readily detectable in field studies [10].
- 50 4. Loss of soil organic matter over decades of salting was so great that
51 immediately beside the road organic matter was no longer substantially
52 solubilized by the high salt concentrations found in soil solution below road
53 drains [9, 11].

54 It might be expected, however, that such impacts would be much less in roadside soils with
55 higher biogeochemical mineral weathering rates (thus a naturally higher pH). The relative
56 contribution of calcium to the total cation pool would naturally be much higher, making the
57 impact of seasonal high sodium inputs less dominant. Incoming sodium ions are energetically
58 unfavorable to displace divalent calcium ions compared to those of hydrogen on the CEC.

59 Therefore, the effects observed on acidic upland soils are likely to be dampened on calcareous
60 soils. This hypothesis has been tested for spray-affected roadside soils at a second sampling
61 site, using a similar protocol and methodology.

62

63 **2. Materials and methods**

64 **2.1 Field site**

65 The study site was an upland area along the A686 at Leadgate, Alston, Cumbria, UK (NY 688
66 430 GB Grid; Lat: 54.780929N Long: 2.486611W), with altitudes up to 609 m above sea level
67 (Fig. 1). The road section used runs parallel to the Black Burn; however, the river could not be
68 sampled at this site due to health and safety implications (especially in winter). The A686 is a
69 priority two route and has a speed limit of 60 mph. In terms of traffic density, the A686 has
70 927 vehicles eastbound and 996 westbound per day (John Robinson, Cumbria County Council,
71 pers. comm.). During the sampling period (October 2005-July 2006), 114 salting outings were
72 completed at a rate of 10-40 g m⁻², depending on the predicted conditions and previous salting
73 regime applied (John Robinson, pers. comm.).

74 The soils are brown earths (Cambisols), with a minimal litter layer but varying degrees
75 of gleying in places. The bedrock is primarily limestone and sandstone. The vegetation is
76 predominately grasses, Juncus and bryophytes and the land is grazed by sheep. Two possible
77 transect scenarios were studied; (a) spray-affected soils (A1 and A2 – south of the A686
78 between the milestone and Blackburn Bank) and (b) control soils (A3 and A4) directly above
79 the road; the two scenarios differ in altitude by only 15 m. Both contaminated and control
80 sampling areas are minimally managed, and have evolved from the same parent material. A
81 wall ca. 1 m high separates the road from the salt-affected sampling sites and an identical wall
82 is above the control transects.

83

84 **2.2 Soil sample collection**

85 Soils for the two salting impact scenarios were sampled in duplicate, with a horizontal spacing
86 of at least 10 m. Transects were laid perpendicular to the road; two were immediately down
87 slope of the highway (A1 & A2). The control (un-impacted) transects, referred to as A3 & A4,
88 were above the road, at ~ 100 m from the highway. Soil samples were collected to 10 cm

89 depth with a stainless steel trowel at 2, 4, 8, 16, 32 and 64 m from the wall (1 m away from the
90 road) on the 20th April 2006, soon after a full period of winter salting. Initial soil sampling was
91 completed on the 5th October 2005 for an inter-seasonal comparison of samples prior to the
92 winter maintenance period commencing. This was done to quantify probable maximum and
93 minimum effects throughout a single salting year.

94 The samples, in polyethylene bags, were stored at 2-4 °C prior to chemical analysis.
95 Stones, identifiable vegetation fragments and roots were removed as quickly as possible by
96 careful hand sorting immediately prior to use.

97

98 ***2.3 Soil solution sampling***

99 To ensure sufficient sample was obtained for analyses, soil solutions for each transect were
100 sampled using sets of three MOM Eijkelkamp rhizon soil solution samplers, 10 cm long and
101 2.5 mm in diameter, evacuated with 60-ml syringes. Soil solutions were sampled at 2, 4, 8, 16,
102 32 and 64 m from the walls, over a depth interval of 5-10 cm.

103 Soil solutions were sampled most intensively throughout the winter period. Sampling
104 was conducted from 12/10/2005-08/07/2006. The air was purged from the syringes, and
105 thereafter, they were stored at 2-4 °C prior to chemical analysis of the samples.

106

107 ***2.4 Soil and soil solution analyses***

108 Soil samples were chemically analyzed for exchangeable base cations, pH, KCl-extractable
109 ammonium-N and nitrate-N, loss on ignition (LOI) and cation exchange capacity (CEC). Soil
110 solution was analyzed for major base cations (Na^+ , K^+ , Ca^{2+} and Mg^{2+}), ammonium-N, nitrate-
111 N, chloride, sulphate, DOC and pH. Identical techniques to those outlined in Green *et al.* [10]
112 were used. Performance of all instrumental methods was checked using synthetic reference
113 standards.

114

115 ***2.5 Statistical analysis***

116 Data from paired soil transects were combined to enhance the representation of each transect
117 scenario (a) indirect and (b) control by increasing the sample size, and non-parametric tests
118 applied. The Kruskal-Wallis test was used to determine whether means were significantly
119 different between control and spray-affected transects, and with distance for each scenario.

120 Significance of correlations was tested using Spearman's rank. For all statistical tests,
121 significance was accepted at $\alpha \leq 0.05$. Analyses were performed using SPSS version 11.0.1
122 (2001).

123

124 **3. Results**

125 **3.1 Soil data**

126 Figure 2 shows ammonium acetate-extractable base cations, exchange acidity and KCl-
127 extractable ammonium as a percentage occupation of CEC at 2, 4, 8, 16, 32 and 64 m, plotted
128 on a log scale, disaggregated by transect type (i.e. spray vs. control). The Na % occupancy of
129 CEC for spray-affected and control transects differs significantly at the 1% level (indirect >
130 control). Percentage occupation of the CEC by sodium does not exceed 0.32% for control
131 soils, compared to maximum value of 1.68% for spray-affected transects at 4 m. The
132 percentage contribution for the impacted transects of sodium to the CEC steadily declines from
133 4 to 64 m, but remained well above the levels observed for the control transects.

134 Calcium % occupation of CEC for salt-affected transects is not significantly below that
135 for the control (Fig. 2). The Ca % of CEC for control transects declines slightly with distance
136 along the control transect from 4 to 64 m. In both control and spray-affected soils Ca %
137 occupation remains relatively high, as hypothesized for the more calcareous soil, and is never
138 at values that would cause concern.

139 At 2 to 4 m the Mg % occupation of CEC for the spray-affected soils exceeds that of
140 the controls. Values for transect types tend to converge at 8 m.

141 The K % of CEC for the salt-impacted soils exceeds that in the control soils at all
142 distances from the highway (1 % level). The control generally appears to fluctuate between
143 0.04 - 0.2%, with no significant difference observed with distance. For the spray-affected
144 transects K % declines from 3% at 4 m to 0.44% at 32 m (5 % level).

145 For H^+ % of CEC the two scenarios show no significant difference. At 2 m the H^+ % of
146 CEC for the both impacted and control soils was 65% (Fig. 2). Soil pH for the control soils
147 showed a significant increase in acidity with distance (at the 5% level).

148 It is immediately obvious (Fig. 2) that CEC for the salt-impacted transects contains
149 consistently (and significantly) higher ammonium % than the control transects (1% level);
150 neither scenario shows a significant change in ammonium % of CEC with distance.

151 Figure 3 shows the percentage occupation of CEC by base cations, hydrogen and
152 ammonium at 2, 8 and 32 m on the 15th October 2005; it may be used to assess seasonal
153 differences from the data from April in Fig. 2. There is a slight reduction in Na % of CEC
154 close to the road, ~ 0.2%, on the salt-affected transects compared with the April data, whilst the
155 control maintains a steady and consistent Na % occupation of the CEC.

156 Between 2 and 32 m, Mg % occupation of CEC increases from 2 or 3.7% to just over 5
157 % for control and spray-affected scenarios respectively in October (Fig. 3). Over the same
158 distance Ca % of CEC on the salt-impacted transects increases from 25.2% to 40.5%, whilst Ca
159 % occupation of the control declines from 40 to 27.1%, similar to the decline seen for the
160 control transects in April.

161 The K % of CEC increased in October for the control transect from the April value of <
162 0.5% to 1.5 – 2%. For the salt-impacted transect, K % of CEC showed a lower degree of
163 variability, and was almost consistent at 2 and 8 m between the two time periods. At 32 m the
164 % occupation of K increased from 0.44 to 2.66% however for spray-affected soils.

165 The H⁺ % of CEC generally shows relatively very little overall change in the degree of
166 acidity between April and October (Fig. 3). For the salt-impacted soils there appears to be a
167 decline in acidity with distance from 2 - 32 m (68-52% of CEC); in contrast the H⁺ % for the
168 control appears to increase from 2 - 32 m (56 to 66% of CEC); however, neither trend is
169 significant at the 5% level.

170 Ammonium as a % of CEC was similar in April and October for the control soils over
171 the spatial range studied, but was apparently higher for the salt-affected soils from 2 – 8 m in
172 April. Though consistent, the effect was not however significant. In these soils, similar spatial
173 trends are apparent for both time periods, with sharp increases in % occupation of CEC by
174 ammonium between 32 - 64 m.

175 The spray-affected transects have, on average, a pH at 2 m approximately 0.5 units
176 lower in April 2005 than the control transects (Fig. 4a; 1% level). This corresponds to the
177 higher calcium dominance of the CEC for the control (Fig. 2). Soil pH of the control transects
178 apparently declines from 5.84 to 5.35 over 64 m, whilst the spray-affected soil pH apparently
179 increases from 5.32 to 5.64, but neither trend is significant. In October (Fig. 4b) the spray-
180 affected soils are still the most acidic, with the difference between the scenarios now being
181 significant at the 1% level.

182 Figure 5 shows the concentrations of KCl-extractable nitrate-N for spray-affected and
183 control transect soils in April 2006 and October 2005. There is no significant difference
184 between the spray-affected and control soils with regard to nitrate-N content for either time
185 period. The spatial variation generally exceeded variation between transects.

186 Figure 6 shows the spatial variation in soil LOI (%) for April 2006 and October 2005
187 for the two transect scenarios. Organic matter loss apparently has occurred from salt-impacted
188 soils compared with control soils. Control soils have a significantly higher LOI (1% level) in
189 both April and October, at a generally consistent level. Salt-impacted soils at 64 m possess a
190 significantly higher LOI (1% level) than those at every other distance; LOI% is highly
191 correlated to the soil water content (70.2%) in April 2006, reflecting greater water retention in
192 more organic soils.

193

194 **3.2 Soil solution data**

195 Sodium concentrations as high as 156 mg l^{-1} were observed 2 m from the road at spray-affected
196 transects beside the A686, but values varied with the prevailing weather conditions. The
197 sodium concentrations for the spray-impacted transects declined to $5\text{-}15 \text{ mg l}^{-1}$ at 64 m down
198 slope; the controls gave a background level of 4.4 mg l^{-1} across all distances. The observed
199 trends in sodium ion concentrations were also apparent in chloride data (significantly
200 correlated to Na^+ at the 1% level). However, whereas at the A6 site Na^+ was largely
201 displacing H^+ , so soil pH increased substantially, at the A686 site there is also strong
202 competition from Ca^{2+} from biogeochemical weathering. Therefore the more strongly retained
203 divalent Ca^{2+} ions have a bigger effect on the composition of the cations occupying the CEC,
204 and the $\text{Ca}^{2+}:\text{Na}^+$ ratio in soil solution is much higher for spray-affected soils at the A686 than
205 it was at the same time of year (April) at the A6 site. This can be very clearly seen in Fig. 7
206 (note the change in scales between sites).

207 Calcium concentrations as high as 86 mg l^{-1} were observed in soil solutions at (data not
208 shown) 4 m from the spray-affected transects. They ranged between $5\text{-}12 \text{ mg l}^{-1}$ for spray-
209 affected transects between 8-16 m. The controls on average had an elevated concentration at 2
210 - 4 m (from $13\text{-}25 \text{ mg l}^{-1}$), thereafter declining and stabilizing from 8-64 m at $10\text{-}12 \text{ mg l}^{-1}$,
211 suggesting a significant Ca^{2+} input from the walls by weathering.

212 Magnesium concentrations were less than 2.5 mg l^{-1} for control transects. Spray-

213 affected transects had the highest Mg^{2+} concentration at 4 m, $\sim 35 \text{ mg l}^{-1}$, and thereafter Mg^{2+}
214 concentration generally declined.

215 Soil solution pH ranged from 6 - 7.5 for spray-affected and 6 - 6.6 for the control
216 transects. As the distance from the highway increased the pH for A1 and A2 tended to decline,
217 whilst A3 and A4 also showed a tendency towards pH decline as distance from the wall
218 increased (Fig. 8). However these trends were nothing like as marked as those seen at the A6
219 site, where the increase in pH attributable to salting was much more consistently around 1 to
220 1.5 pH units. These trends at the A686 site probably reflect the impact of the limestone walls.

221 In the earlier study at the A6 site, the soil solutions from the spray-affected transects
222 consistently contained much higher (more than an order of magnitude) nitrate concentrations
223 than the control soil solutions. This resulted in substantially increases in the nitrate
224 concentration in a local, adjacent river [8]. The control soil solutions in the A686 study also
225 contained minimal amounts of nitrate-N (generally $< 1 \text{ mg l}^{-1}$), whilst the salt-affected transects
226 had spatially and temporally variable concentrations (Fig. 9; note change in scale). Nitrate-N
227 concentrations on spray-affected transects showed no consistent trend attributable to a road
228 salting effect, although 9 out of 96 samples contained a concentration $> 5 \text{ mg nitrate-N l}^{-1}$.

229 At 2 - 4 m, and (more consistently) at 64 m, ammonium-N concentrations apparently
230 tended to be higher for the salt-affected transect soil solutions (Fig. 10). This might have been
231 anticipated from the apparent increase in ammonium % of CEC for the salt-affected transects,
232 but that effect was not significant, as discussed earlier. Ammonium-N concentrations in control
233 soil solutions were very low, mostly below 0.2 mg l^{-1} . For salt-affected transect A1 (but not
234 for transect A2), ammonium-N concentration was consistently high from 2 to 16 m on
235 08/07/2006, but for no obvious reason.

236

237 **4. Discussion**

238 Some sodium ion effects are clear from the differences between the spray-affected and control
239 transects. Figure 2 demonstrates the percentage occupation of cation exchange sites for
240 sodium, magnesium, calcium, potassium, ammonium-N and hydrogen ions in April 2005.
241 There is clear sodium elevation for the spray-affected transects as compared to the control over
242 2 - 64 m. However the contribution of Na^+ to exchangeable base cations is lower than at the
243 A6 site [7]. The higher Na % occupation of CEC does not directly correspond to a lower H^+ %

244 of CEC for the control transects, and thus a lower H^+ % of CEC for spray-contaminated
245 transects at the A686 site, in marked contrast to observations for the A6 site [7]. The H^+ % of
246 CEC is inversely correlated with the % Ca and Mg of CEC (1% level) at the A686 site. Thus,
247 it is more likely that the exchangeable hydrogen ions are competitively displaced by incoming
248 calcium and magnesium ions than by sodium. Evidence presented earlier suggests this is
249 partly from the limestone walls above all transects at the site.

250 Green and Cresser [7] hypothesized that close to the highway a reduction would be
251 observed in the calcium and magnesium % occupation of the CEC in the acid soils of the A6
252 site, but the opposite was observed in practice. It was thought that the higher levels of calcium
253 were a result of the calcium content in the rock salt [7, 10]. Hence, it would not be surprising
254 to observe the same trend at the A686 as a result of the same road salt product being used.
255 However, Ca and Mg % of CEC values are elevated above those observed at the A6 for the
256 spray-affected and control sites, suggesting additional source(s) of calcium and magnesium.
257 This is also reflected in the raised calcium concentrations in soil solutions for the controls.

258 It is clear that the calcium input via limestone dissolution (parent material or wall
259 components) is having an affect on pH and Ca % on the CEC. The Ca % of CEC is greater for
260 the A686 at 2 m (26.6%) than for the A6 (17.7%) in April. This is also apparent in October for
261 the spray-affected transects. It is thought that the additional calcium from the limestone wall
262 constructed above both the control and spray-affected transects is very significant because pH
263 declines with distance for the control transect. Mobile anion effects contribute to the lower soil
264 solution pH for the spray-affected transects. The sodium input effects on pH are effectively
265 negligible at the A686 sites.

266 Initially there appeared to be some evidence to suggest that the N-cycle is being
267 disrupted by the use of road salts at the A686 site. However, spatial and temporal shifts in pH
268 will not be modifying mineralization and nitrification rates significantly, so none of the
269 apparent effects on the N cycle were significant statistically. Green and Cresser [7, 8] showed,
270 in contrast, that displacement of ammonium ions from the exchange sites occurred in reaction
271 to increased sodium concentrations during salting of the A6. It is interesting to note that the
272 percentage CEC occupation by ammonium observed at the A686 is consistent with that of the
273 spray-affected soils at the A6.

274 Spray from road salting had a very strong and significant effect on soil solution nitrate

275 concentration in the naturally acidic soils alongside the A6 [8]. Figure 9 showed that no such
276 strongly consistent significant effect occurred at the A686 site. This is almost certainly
277 because of the lack of consistent significant pH change at the A686 site.

278 LOI is, as for the A6 soils [8], greater for the control soils than for the spray-affected
279 soils (Fig. 6 - 1% level). In comparison to the A6, however, there does not appear to be a build
280 up of organic matter down slope from organic matter dispersal by high Na^+ concentrations.
281 This may be is a product of topography (lesser degree of slope angle for the A686). It seems
282 most likely that at the A686 site the dominant cause is roadside pollution of diverse types
283 lowering the annual input of organic litter to the roadside soils.

284 The A686 shows signs of reversibility of salt effects there after road salt application
285 had ceased for the season, *via* reduction in sodium ion % of CEC between April and October.

286 The acidic upland site of the A6 possesses a host of ecological implications through the
287 introduction of elevated concentrations of road salt (cation exchange reactions) and the
288 significant temporal/spatial pH shift. Changes in pH lead to changes in microbial activity
289 which is highlighted by the enhanced nitrification observed at the A6, which have a
290 reverberation on the N cycle. Displacement of ammonium ions from the CEC may lead to N
291 deficiencies in roadside soils. In addition toxicity effects of high salinity on microbes may be
292 masked by the signal observed. Vegetation may be affected by road salt in several ways: by
293 accumulating Na^+ in toxic concentration within plant tissues (Townsend, 1980); by increasing
294 osmotic pressure differences causing desiccation (drying) (Bryson & Barker, 2002; Larcher,
295 1995; Townsend, 1980); by altering mineral nutrition balances (reduced levels of available
296 ammonium, and K^+ within the soil) (Bayuelo-Jiménez *et al.*, 2003; Serrano & Rodriguez-
297 Navarro, 2001); and by altering soil structure. However, damage usually results from a
298 specific ion effect rather than from an increase in osmotic potential (Townsend, 1980). The
299 specific ion effect involves movement of the ions such as sodium and chloride into plant cells,
300 where they adversely affect cell membrane stability, metabolism and growth, and at high
301 concentrations is toxic, possibly leading to fatality (Larcher, 1995; Townsend, 1980). In
302 addition ammonium and nitrate leaching may also suppress vegetal growth.

303 Furthermore, elevated chloride concentrations are known to interfere with
304 photosynthesis in algae in surface waters (Williams *et al.*, 1999), with shifts in population
305 occurring with 12–235 mg l^{-1} due to the varying degree of sensitivity between algae species

306 (DOE & DOH, 2001). In higher organisms they can result in potentially fatal metabolic
307 acidosis and osmotic stress, as well as behavioural changes (Williams *et al.*, 1999). It is
308 possible that changes in population, community structure and/or biodiversity may occur due to
309 acute and chronic toxicity of road salts, in combination with other physical and chemical
310 impacts generated directly or indirectly from the application of road salt.

311 Raised levels of sodium ions in water can cause high blood pressure and hypertension,
312 so individuals whom already suffer from this condition and on salt-restricted diets should not
313 ingest greater than 20 mg l⁻¹. Several states in the northeast US and Canada have measured
314 sodium from road salt in well waters at concentrations that are 2-140 times the recommended
315 limit for individuals on salt-restricted diets (Amrhein *et al.*, 1992). Hence, increases in sodium
316 and chloride ions can also cause problems with water balance in the human body. This is a
317 particular concern when considering upland soils and associated freshwater bodies as these
318 tend to be used as potable sources in the UK (Smart *et al.*, 2001).

319 In addition enhanced loadings of nitrate and ammonium may occur as product of cation
320 displacement and enhanced nitrification as a result of the effect of pH on the soil microbe
321 community, both of which have an eutrophication risk.

322 In the case of the A686 such implications are unlikely to be realized due to the presence
323 of the higher rate of weathering of parent material. The occupation of the CEC with Ca²⁺ and
324 Mg²⁺ in the first instance protects the soil ecosystem for a shift in pH due to the displacement
325 of H⁺ ions. This effectively protects the roadside environment to the long-term episodic
326 influxes of road salt. Furthermore, such ecological implications outlined above may only occur
327 in extreme cases.

328

329 **5. Conclusion**

330 The data from the A6 and A686 spray-affected transects, compared with the relevant control
331 transects, conform well only in terms of Na⁺ % occupation of the CEC. The high weathering
332 rate of the calcareous parent material within the wall and in the soil itself, rather than high salt
333 input, regulates the soil pH down-slope at the A686 site. As a consequence at the A686 site the
334 disruption of the N cycle is negligible compared with that reported earlier for the A6 sites. It
335 may be concluded that major N cycle disruption that occurs as a consequence of road salting
336 when roads run through naturally very acidic soil ecosystems is much less of an issue when the

337 roadside soil ecosystems are calcareous. That said, the lower organic matter content in salt
338 impacted soils of both types (indicated by significantly lower LOI values) warrants further
339 investigation.

340

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344

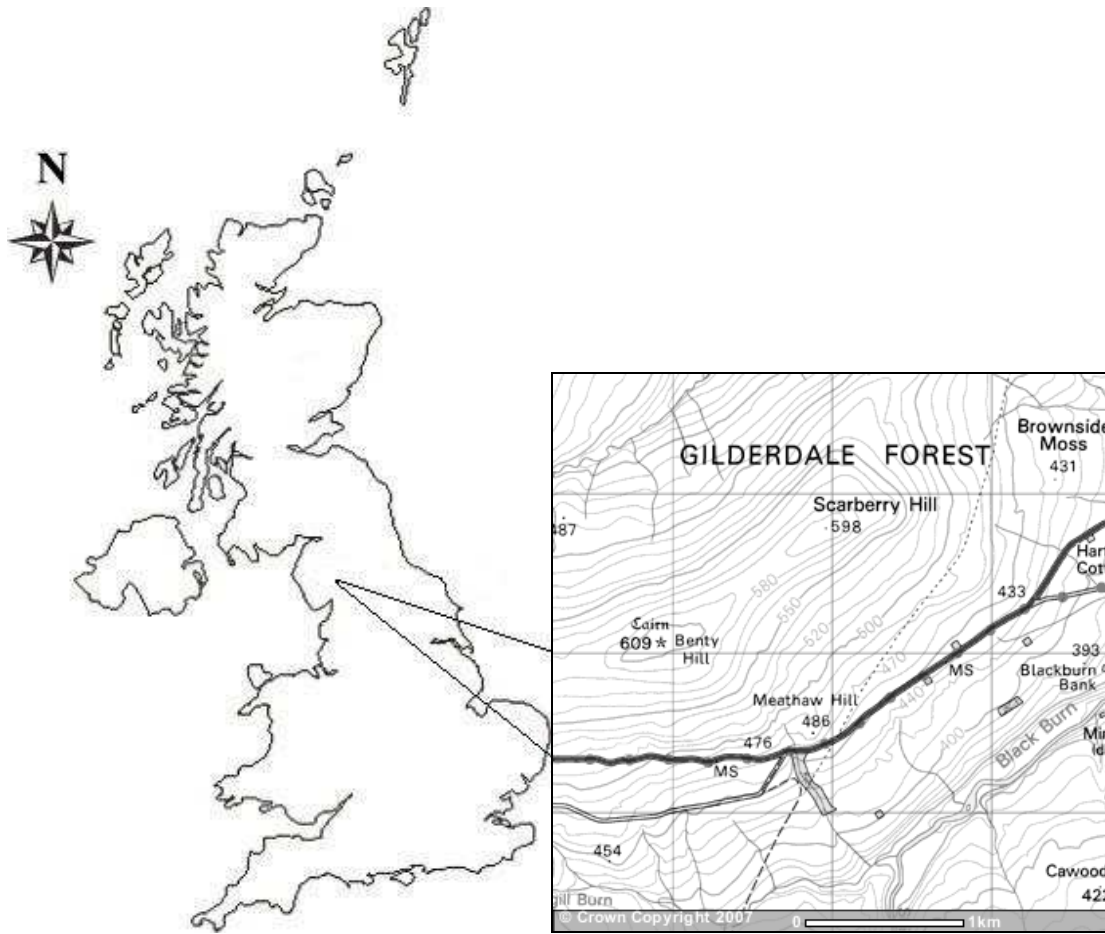
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379 Figure 1. Map showing the location of the A686 field site, Leadgate, Alston, UK. (C)
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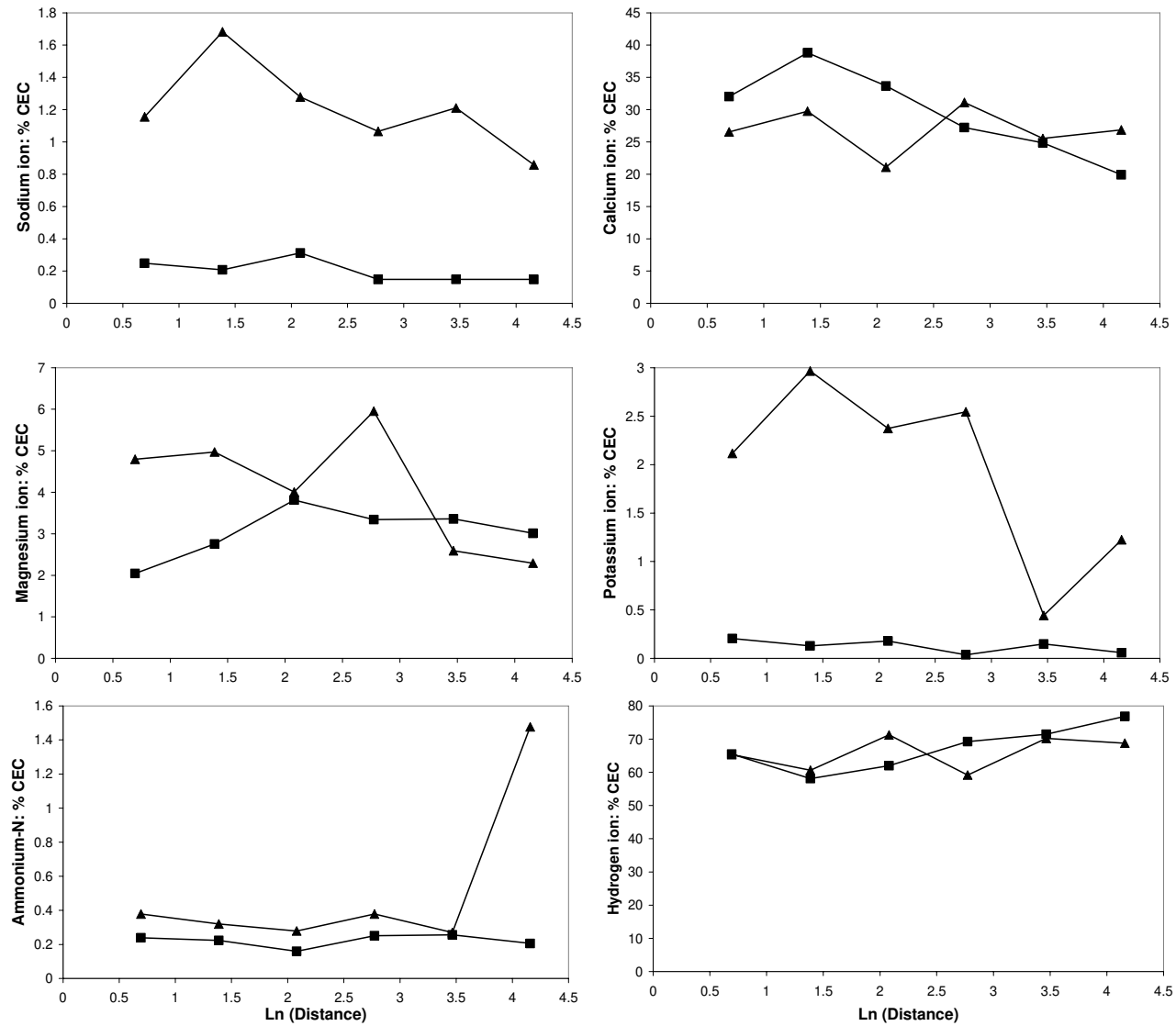


Figure 2. Proportion of base cations, ammonium-N and hydrogen of CEC (%), April 2006 at 2, 4, 8, 16, 32 and 64 m away from the wall for spray-affected (triangles) and control transects (squares).

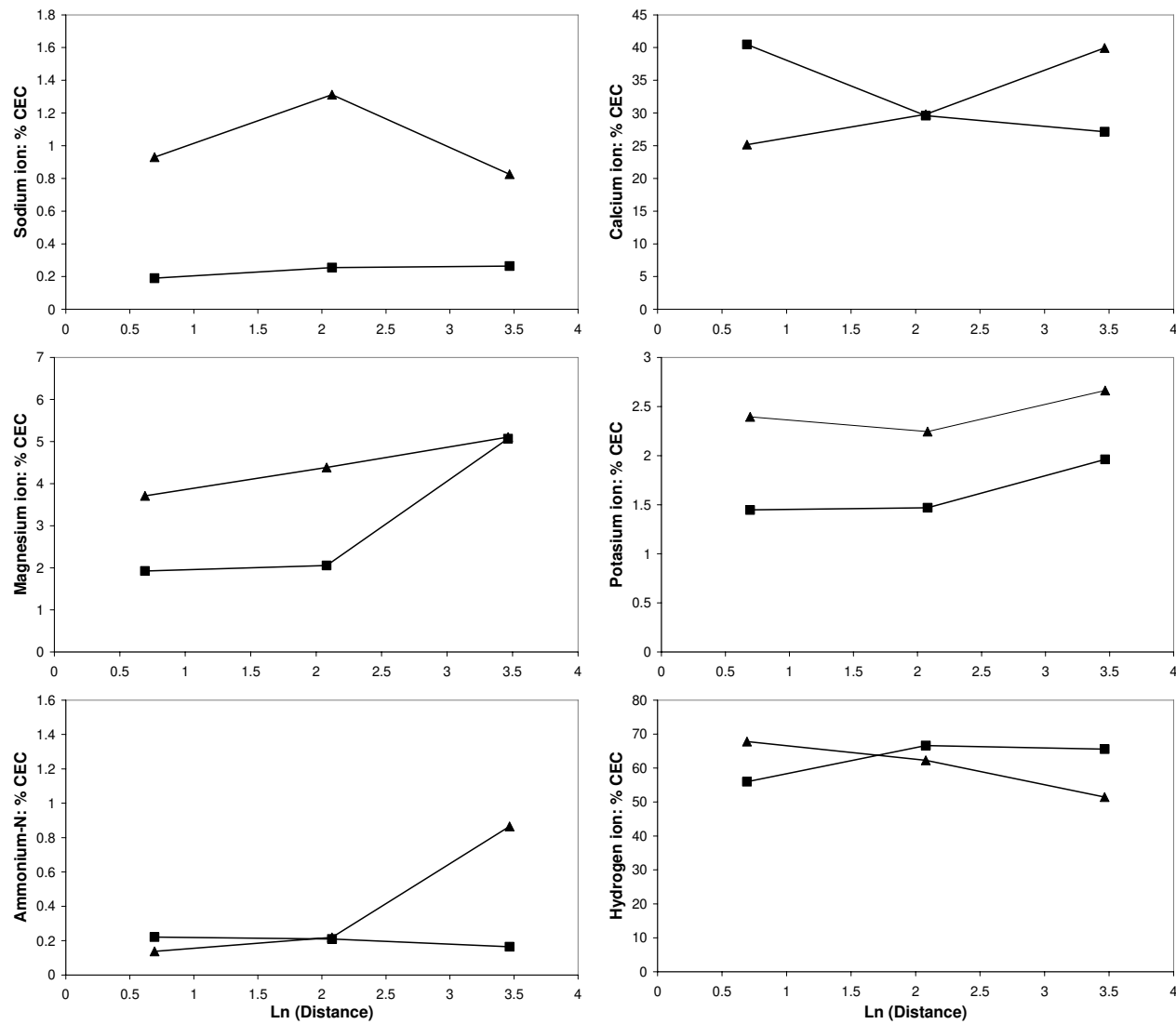


Figure 3. Proportion of base cations, ammonium-N and hydrogen on the CEC, October 2005 (%) at 2, 8 and 32 m from the wall, for spray-affected (triangles) and control transects (squares).

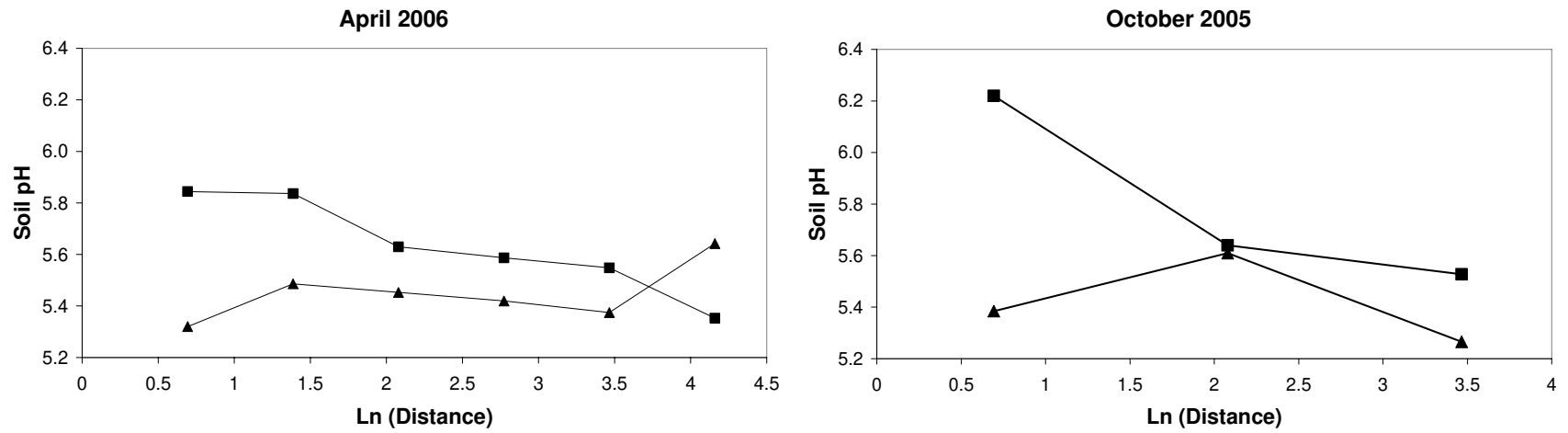


Figure 4. Soil pH (H₂O) for April 2006 and October 2005 for soil samples at 2, 4, 8, 16, 32 and 64 m from the wall in April and 2, 8 and 32 m in October, for spray-affected (triangles), and control (squares) soils.

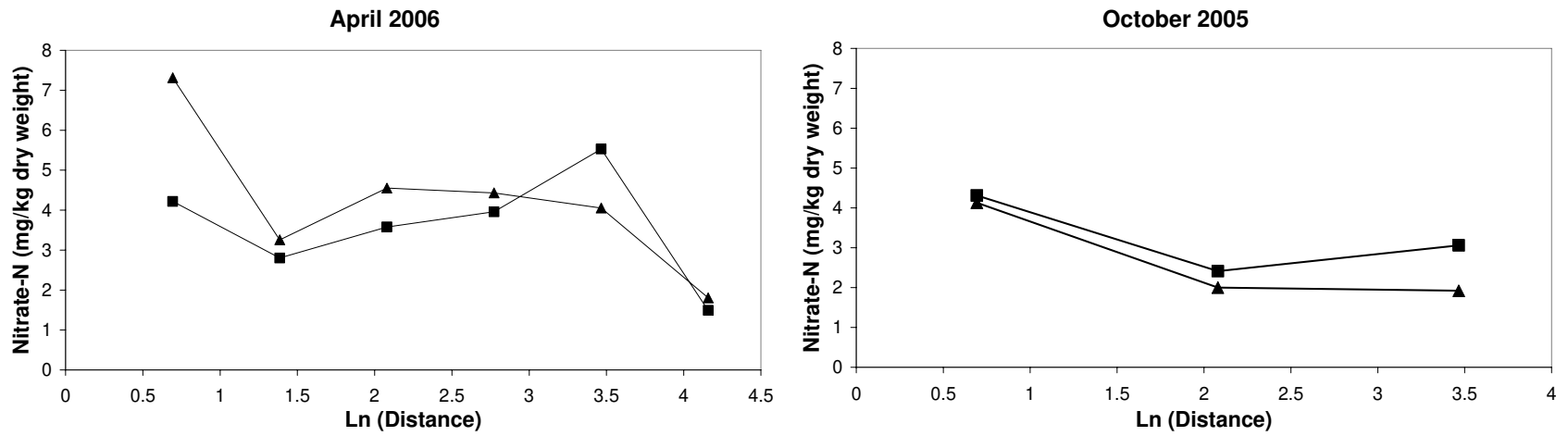


Figure 5. KCl-extractable nitrate-N concentration (mg/kg) for April 2006 and October 2005 for soils sampled at 2, 4, 8, 16, 32 and 64 m from the wall in April and 2, 8 and 32 m in October, for spray-affected (triangles) and control (squares) soils.

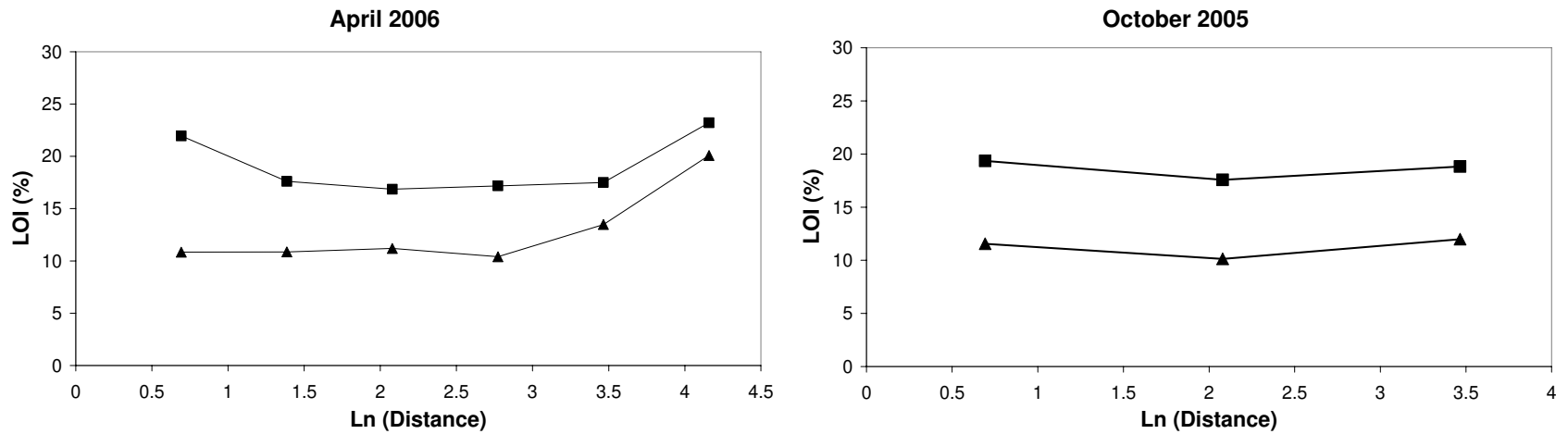


Figure 6. Loss-on-ignition (%) for April 2006 for soils sampled at 2, 4, 8, 16, 32 and 64 m from the wall (April) or 2, 8 and 32 m (October) for salt-affected (triangles) or control (squares) soils.

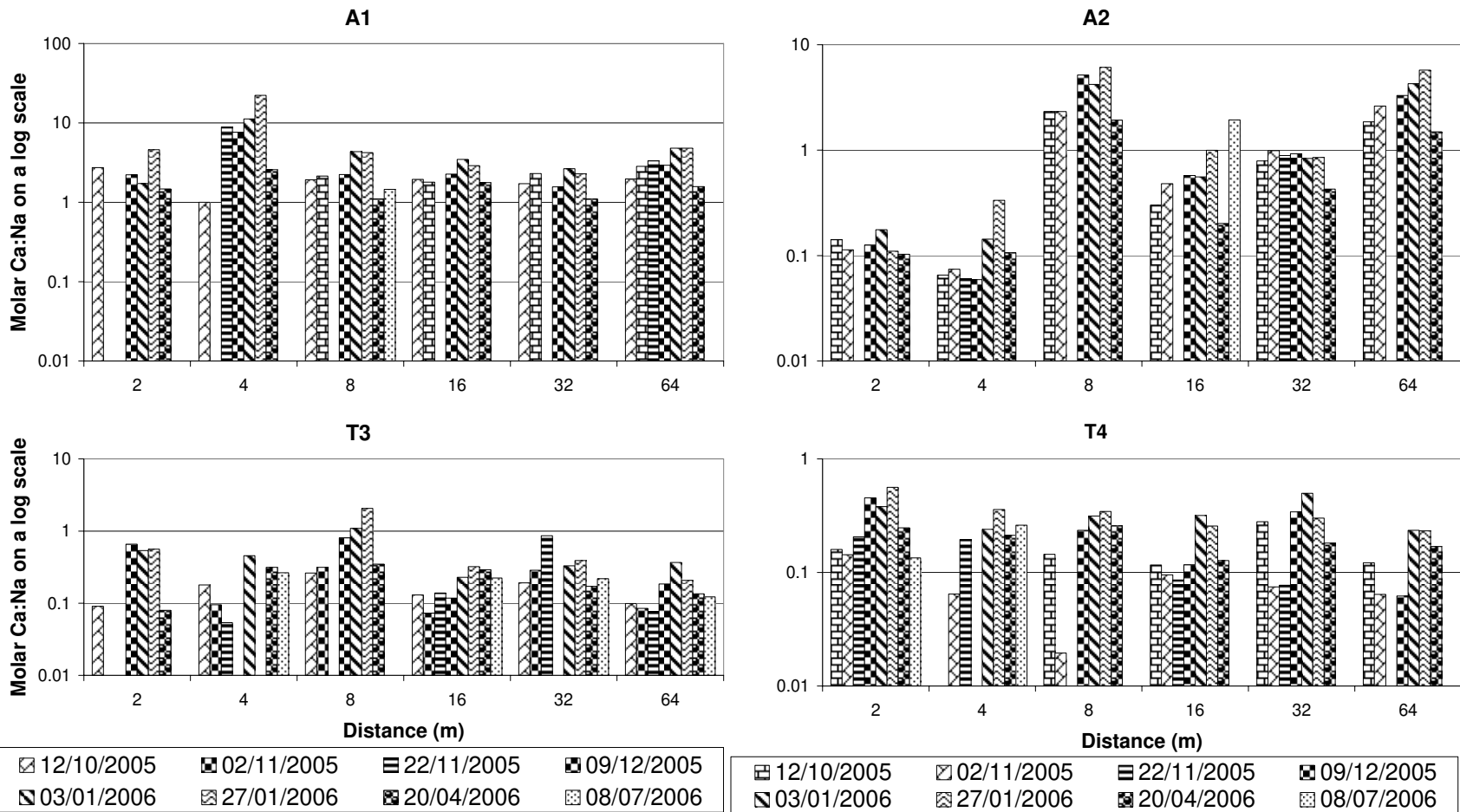


Figure 7. Comparison of the ratio of calcium to sodium concentrations, expressed on a mol_c basis on a log scale, in soil solutions along the lengths of the spray effect transects in this study (A1 and A2, top) and in the earlier A6 study (T3 and T4, bottom). Note changes in scale.

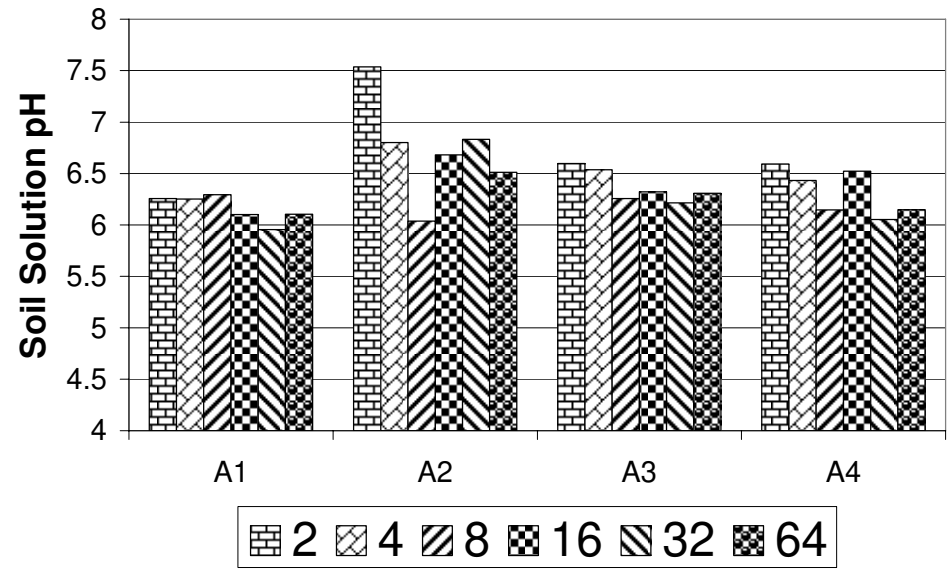


Figure 8. Soil solution pH at 2, 4, 8, 16, 32 and 64 m for spray-affected soils (A1 & A2), and control soils (A3 & A4).

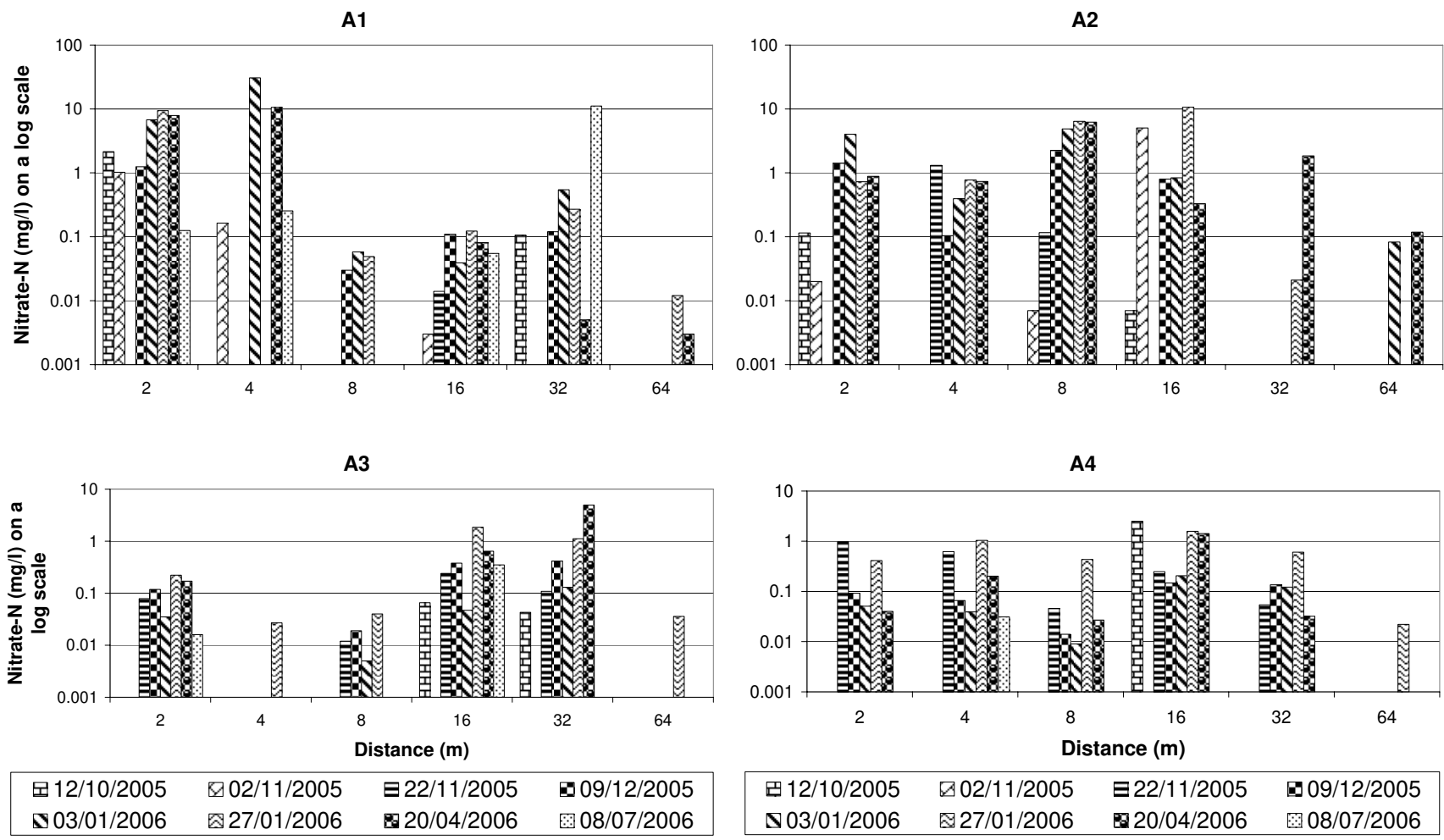


Figure 9. Nitrate-N concentration on a log scale for soil solutions at 2, 4, 8, 16, 32 and 64 m for spray-affected soils (A1 & A2) and control soils (A3 & A4). Note changes in scale.

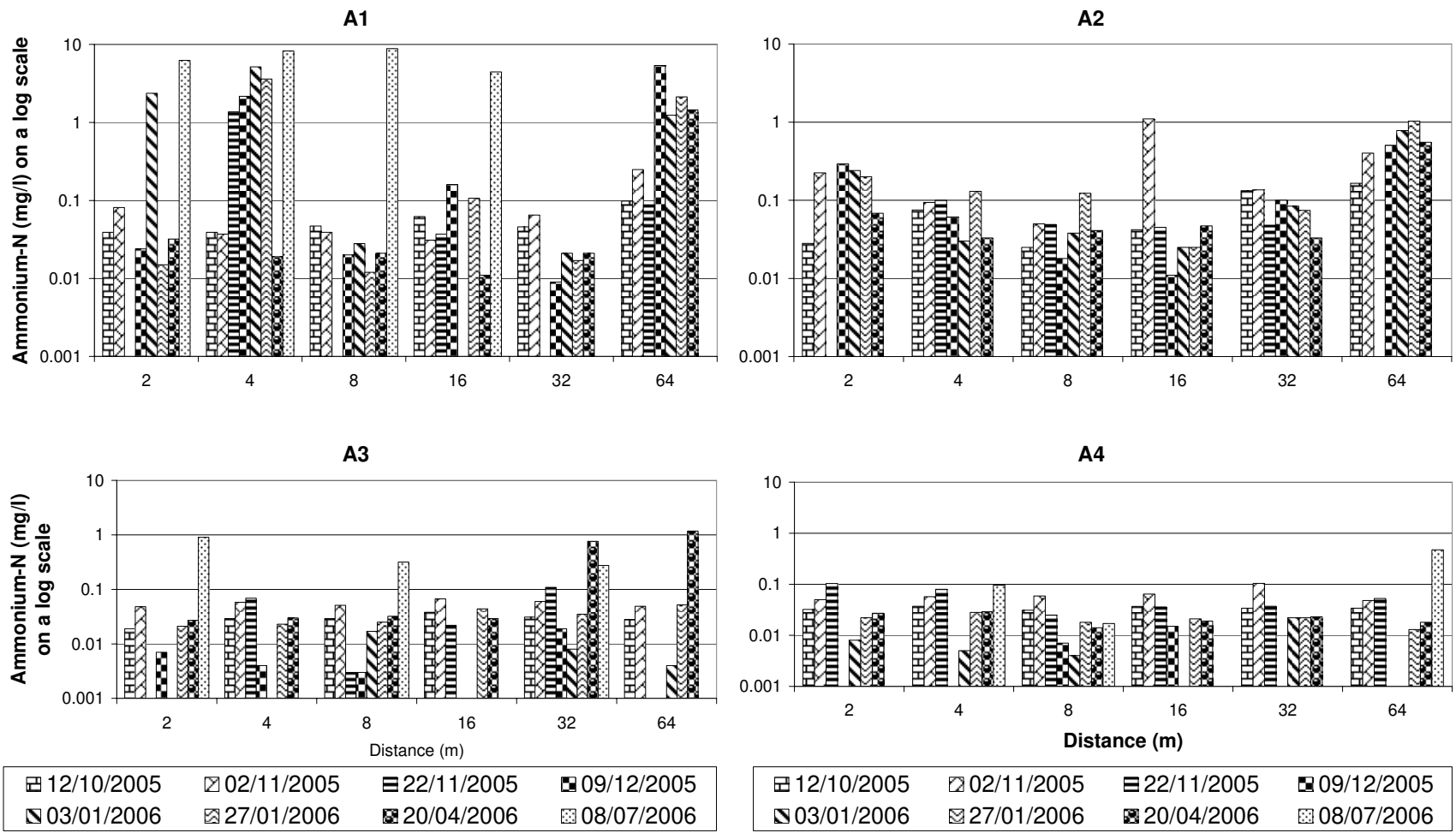


Figure 10. Ammonium-N concentration on a log scale for soil solutions at 2, 4, 8, 16, 32 and 64 m for spray-affected soils (A1 & A2), and control soils (A3 & A4).