NUTRIENT DYNAMICS IN A PEATLAND FOREST RIPARIAN BUFFER ZONE AND IMPLICATIONS FOR THE ESTABLISHMENT OF PLANTED SAPLINGS

J. FINNEGAN[a], J.T. REGAN[a], E. de EYTO[b], E. RYDER[c], D. TIERNAN[d] and M.G. HEALY[a]*

[a]Civil Engineering, National University of Ireland, County Galway, Ireland.

[b]Marine Institute, Newport, County Mayo, Ireland.

[c]Centre for Freshwater Studies and Department of Applied Sciences, Dundalk Institute of Technology, Dundalk, Co. Louth, Ireland.

[d]Coillte, Cedar House, Moneen Road, Castlebar, Co Mayo, Ireland.

* Tel +353 91 495364 fax +353 91 494507, email mark.healy@nuigalway.ie

ABSTRACT

Forestry on peatland throughout the world is now focused on minimising destructive effects to the surrounding environment, especially during harvesting. These effects may be mitigated through the use of well-developed riparian buffers zones (RBZs). However, much of the commercial forestry planted in Ireland and the UK in the mid 20th century was planted without adequate RBZs. The creation of new RBZs prior to clearfelling may be a possible mitigation measure in these circumstances. The aim of this paper was to assess the nutrient content and phosphorus (P) adsorption capacity of the soil, and survival of planted saplings in

30   a RBZ, positioned downslope from a standing forest and partly covered with brash mats, five

31   years after its establishment. Dissolved reactive phosphorus (DRP) concentrations were

32   significantly higher under the brash mats in the RBZ when compared to all other areas. The

33   standing forest had the highest concentrations of ammonium nitrogen (NH_4–N), while total

34   oxidised nitrogen (TON) was similar for all areas. Water extractable phosphorus and

35   desorption-adsorption testing also confirmed the high concentrations of P under the brash

36   mats, but P did not leach through the peat to the stream. The overall survival rate of the

37   saplings was relatively high, with over half of *Quercus robur* (oak) (57 %), *Sorbus aucuparia*

38   (rowan) (57 %) and *Betula pendula* (birch) (51 %) surviving. *Salix cinerea* (willow) (22 %),

39   *Alnus glutinosa* (alder) (25 %) and *Ilex aquifolium* (holly) (44 %) did not survive as

40   successfully. The RBZ was capable of providing nutrients for the survival of planted saplings,

41   fertilizing the peat with degrading brash material and preventing elevated levels of nutrients

42   entering the adjacent aquatic ecosystem.

43

1. Introduction

Peatlands are found in over 175 countries worldwide, are mostly present in moist temperate climates in the northern hemisphere (Sjörs, 1980), and cover approximately 3 % of the total landmass in the world (4,000,000 km^2) (Bain et al., 2011). These ecosystems produce 10 % of the global freshwater supply and one-third of the world's soil carbon content (Joosten and Clarke, 2002). Approximately 150,000 km^2 of this landmass has been drained for commercial forestry, while the area not commercially drained, but forested, is unknown (Joosten and Clarke, 2002). Ireland's forest cover stands at 10.15 %, or 700,000 ha, of the total surface area of the island (National Forest Inventory, 2007). The Irish State, under the management of the Forest Service, carried out the majority of the afforestation in the mid 20th century. This was mainly coniferous plantation on non-productive agricultural land (Bacon, 2003). It is estimated that 59.6 % (417,200 ha) of forestry in Ireland is on peat (National Forest Inventory, 2007) and approximately 300,000 ha of afforestation is on upland peat areas (EEA, 2004; Rodgers et al., 2010). Harvesting of forestry on peat can be challenging due to high soil water contents (gravimetric water contents usually exceed 800 % (Long and Jennings, 2006)), low ground bearing capacities of between 10 and 60 kPa (Owende et al., 2002) and the vulnerable nature of the ecosystem (Forest Research, 2009). In Ireland, forestry harvesting practice (including thinning) minimises soil disturbance by adopting appropriate mitigation measures such as: (1) the use of low ground pressure machines and (2) the laying of brash mats, consisting of small branches and logs under all paths used by the felling and extraction machinery. The scale of soil disturbance to a clearfell site is dependant on a combination of factors,

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69   including the number of passes by machinery, soil water content and the effective use

70   of brash mats (Gerasimov and Katarov, 2010).

71

72   Forestry on peatland throughout the world is now moving towards a 'progressive

73   management approach' (Joosten and Clarke, 2002), which incorporates sustainable

74   timber production alongside multiple uses such as habitat restoration, ecological

75   regeneration and the minimisation of any potentially negative effects to the

76   surrounding environment. These negative effects may include eutrophication (an

77   increase in nutrient levels in a watercourse causing excessive flora growth (Sharpley et

78   al., 2003)), sedimentation (an increase in suspended sediment (SS) release to a

79   watercourse causing damage to water ecology (Rodgers et al., 2011)) and biodiversity

80   loss (a change of species, genetic and ecosystem diversity (Walker, 1992)). Coillte, the

81   Irish State's current forest management company, is certified under the Forest

82   Stewardship Council (FSC) to enforce strict environmental, economic and social

83   criteria for sustainable forest management (Coillte, 2012). This progressive and

84   sustainable management approach includes more effective planning to provide

85   protection to water-courses from drainage, fertilisation and afforestation, final harvest

86   and regeneration (Owende et al., 2002). Some of this protection may be provided by

87   riparian buffer zones (RBZs).

88

89   The standard forestry practice in Ireland and the UK at the time of afforestation (in the

90   1950s) led to trees being planted in areas adjacent to water-courses with no allowance

91   for a RBZ (Broadmeadow and Nisbet, 2004; Ryder et al., 2011). This lack of a buffer

92   may result in elevated nutrient and SS release into water-courses during clearfelling

93   (Carling et al., 2001). Other negative effects in the absence of RBZs are the excessive

94   quantity of shade to the stream provided by the overhanging mature conifer

95   plantations, which leads to a death of the riparian vegetation and leaves the bank sides

96   susceptible to erosion (Broadmeadow and Nisbet, 2004). The presence of commercial

97   conifers close to the edge of a stream is also likely to affect the emergence of

98   invertebrates and the biodiversity in comparison to deciduous trees (Broadmeadow and

99   Nisbet, 2004; Kominoski et al., 2012). Much of the commercial coniferous forestry

100   planted in the 1950s is now at harvesting age and the adoption of current forest

101   practice which creates RBZs will minimise the risk of negative impacts on receiving

102   waters for successive rotation.

103

104   Riparian buffer zones are used in forestry worldwide in areas such as Fennoscandia

105   (Syversen and Borch, 2005; Väänänen et al., 2008), the USA and Canada (Aust and

106   Blinn, 2004; Luke et al., 2007), and in New Zealand (Parkyn et al., 2005), to

107   ameliorate the negative impacts of forestry on adjacent water-courses. In the UK,

108   forestry planning since the 1990s has allowed for RBZs of native hardwoods to

109   provide shade and shelter for wildlife and the stream inhabitants, and for existing

110   conifer streamside plantations to be felled and restored (Farmer and Nisbet, 2004).

111   Current forest practice in Ireland incorporates the use of buffer zones along waterways,

112   with widths of between 10 m and 25 m depending on slope and soil erodibility (Forest

113   Service, 2000). However, RBZs need to be created in old forest stands on peat soil in

114   the most sustainable method possible.

115

116   A RBZ can be created in two ways: (1) by leaving an intact strip of forest adjacent to

117   the stream and clearfelling the main coupe of trees behind it, or (2) by harvesting the

118   trees from a strip beside the stream a number of years prior to clearfelling the main

119   coupe and allowing the area to revegetate, either naturally or artificially (Ryder et al.,

120   2011). Forest buffer zones (option one, with trees left in buffer zone) in the UK have

121   been shown to be successful at allowing sedimentation to occur within the buffer

122   because of a slowing down of the surface runoff due to the well-structured and

123   normally drier character of forest soils (Broadmeadow and Nisbet, 2004) and the

124   increased macroposity from tree roots and soil fauna (Goudie, 2006). This is coupled

125   with the damming effect created by falling debris and protruding roots in the forest

126   buffer, which form sediment traps (Broadmeadow and Nisbet, 2004). However, this

127   option may not be practical in the west of Ireland due to thin soil depths, exposed sites

128   and high winds, leading to the increased chance of wind throw close to the

129   watercourse, resulting in a higher risk of sedimentation and nutrient runoff. The second

130   RBZ creation option has potential to be adopted in Ireland, as it increases the primary

131   production in the stream, provides adequate shade and leaf litter, promotes greater

132   biodiversity and taxon richness, and increases sunlight to the watercourse (Ryder et al.,

133   2011). Ground vegetation is also an important method of slowing down flow and

134   trapping sediment (Broadmeadow and Nisbet, 2004). It has been noted, however, that

135   there can be a significant time delay in establishing ground vegetation on sites on

136   which the logging residues have been left (Broadmeadow and Nisbet, 2004). Ormerod

137   et al. (1993) conducted a study on 11 upland streams in forestry catchments that had

138   been clearfelled from one to seven years prior to their study and noted that the streams

139   had retained some of the characteristics of a forestry catchment stream, even after 7

140   years of recovery. Ryder et al. (2011) found that the creation of RBZs resulted in

141   increased water discharge and significantly higher SS loads to receiving waters, an

142 elevated stream temperature, and minor changes in the average abundances and taxon

143 richness of macroinvertebrate communities. These effects were consistent with the

144 short-term negative impacts of felling at the time of creation of the RBZs.

145 Nevertheless, the creation of RBZs in this way was not felt to have catastrophic effects

146 on the receiving water course and its inhabitants, and any impacts were short-lived

147 (Ryder et al, 2011).

148

149 Coillte's District Strategic Plan 2011 – 2015 specifies a 20 m unplanted strip followed

150 by 10 – 20 m of broadleaf plantation between a permanent water-course and conifer

151 forest (Coillte, 2011). This would result in the production of scrub broadleaf cover

152 with a protective function only (Coillte, 2011). The tree species planted in a RBZ are

153 generally recommended to be the native variety and species choice will have an impact

154 on the efficiency of the buffer (Broadmeadow and Nisbet, 2004). Factors such as shade

155 and canopy density need to be taken into consideration, as RBZs are seen to function

156 more efficiently when there is a high level of ground covering plants (Broadmeadow

157 and Nisbet, 2004). Dense planting of species with larger leaf areas, like *Alnus* (alder)

158 or *Quercus* (oak), may provide too much shade for the successful growth of the lower

159 ground covering plants and it is recommended that they are not planted in large groups,

160 but rather dispersed throughout the RBZ with species with lower canopy density such

161 as *Salix* (willow), *Betula (*birch) and *Sorbus* (rowan) (Broadmeadow and Nisbet,

162 2004). Alder is also suspected of adding to stream acidification due to its ability to fix

163 nitrogen (N) from the atmosphere and, therefore, should be limited in RBZ

164 regeneration projects (Broadmeadow and Nisbet, 2004). It is relatively unknown which

165 (if any) native deciduous species are likely to survive, if planted in upland peats

166 following clearfelling of coniferous forest.

167

168 Due to the upland nature of these areas, many of these catchments include headwater

169 streams, which are important salmonid habitats and need to be protected from nutrient

170 enrichment. The phosphorus (P) retention capacity of a soil is partly dependant on its

171 abundance of aluminium (Al) and iron (Fe) compounds (Giesler et al., 2005; Väänänen

172 et al., 2006). Aluminium and Fe are readily available in mineral soil, but are lacking in

173 peat. However, as the mineral layers, where they occur, in riparian peatland buffers aid

174 in retaining higher quantities of P than peat further back from the riparian zone

175 (Väänänen et al., 2006), one option to mitigate P loss from peat forests to receiving

176 waters is to create RBZs in existing forest stands prior to clearfelling the main coupe

177 behind the buffer zone (Ryder et al., 2011). Desorption-adsorption isotherms can

178 indicate the amount of P retained in the soil and show the adsorption properties of the

179 soil, while water extractable phosphorus (WEP) testing measures the readily available

180 fraction of the soil P and is used as an indicator of the amount of P that may be carried

181 from a soil by surface runoff in storm events. Current recommended buffer widths in

182 Ireland of 10 – 25 m may not be capable of removing all nutrients from the runoff

183 during high storm events when the majority of the P is transported, as the retention

184 time may be too short for uptake of soluble P by vegetation (Rodgers et al., 2010). It

185 has been shown that elevated levels of nutrients and sediment are frequently associated

186 with clearfelling operations for up to 4 years (Cummins and Farrell, 2003; Rodgers et

187 al., 2010; Rodgers et al., 2011). Although P can become fixed in the soil and only a

188 small amount may be leached to water-courses (Haygarth et al., 1998), even small

189 concentrations (> 35 µg L^{-1} molydbate reactive phosphorus (MRP)) can have a

190 negative impact on water quality (Bowman, 2009), leading to restrictions of use for

191 fisheries, recreation, industry and drinking water (Elrashidi, 2011; Sharpley et al.,

192 2003). Phosphorus can be found in both dissolved and sediment-bound (minerals and

193 organic matter) forms. Dissolved P is bio-available and is therefore the main cause of

194 eutrophication in freshwater (Elrashidi., 2011; Regan et al., 2010; Sharpley et al.,

195 2003; Väänänen et al., 2006). In Ireland, P is the limiting nutrient for eutrophication

196 (Hutton et al., 2008) and is therefore the nutrient of greatest interest. The limit for

197 MRP, which is similar to dissolved reactive phosphorus (DRP) (Haygarth et al., 1997),

198 concentrations in Irish rivers to maintain 'good ecological status' is 35 µg L^{-1} and for

199 'high ecological status' is 25 µg L^{-1} (Bowman, 2009). A conservative value of 30 µg L^{-1}

200 has been statistically linked with lower biological Q ratings (biological quality

201 ratings) (EPA, 2005), phytoplankton production (Daniel et al., 1998) and increased

202 algal growth in freshwaters (Haygarth et al., 2005).

203

204 The aim of this study was to examine the characteristics of an uncultivated RBZ, in an

205 upland peat area in the west of Ireland. The RBZ was clearfelled 5 yr previous to the

206 present study and restocked 1 yr later with group planted broadleaf species.

207 Specifically, the following characteristics were examined in the 5 yr old RBZ: (1) the

208 deciduous species of trees which were able to survive and thrive (2) the soil and

209 surface water nutrient content, and (3) the P adsorbing capacity of the soil in the

210 regenerated zone and in the standing forest. This allowed for an assessment of the

211 function and performance of RBZs to supply nutrients to various native species of

212   growing saplings within peatland forestry, and to provide some protection against

213   nutrient export into receiving waters.

214

215   **2.   Materials and Methods**

216

217   **2.1.   Study Site Description**

218

219   The study site was located in the Altaconey (also known as Altahoney) forest in the

220   Burrishoole catchment in Co. Mayo, Ireland (ITM reference 495380, 809170) (Figure

221   1). This catchment is situated in the Nephin Beg range at an approximate elevation of

222   135 m above sea level. The study stream is a third-order stream (Strahler, 1957) and is

223   located within a subcatchment area of 416.2 ha, of which 176.4 ha is fully forested

224   (Ryder et al., 2011). The site has a north-westerly aspect, while the study stream,

225   which is one of the main tributaries to the Altaconey River, flows in a southwest-to-

226   northeast direction to the north of the site before turning south to join the Altaconey

227   River. There is a moderate climate, which is heavily influenced by the proximity of the

228   Atlantic Ocean, with average air temperatures of 13 °C in summer and 4 °C in winter.

229   The site is subjected to approximately 2400 mm of rainfall every year, with 289 rain

230   days between May 2010 and April 2011. As a result, the area is characterised by

231   upland spate streams and gorged drains. The Altaconey river responds quickly to

232   rainfall events, and discharge frequency curves are characterised by steep amplitudes

233   and extremely fast falling crests, with 75 % of the total runoff in the Altaconey river

234   originating from direct runoff (Muller, 2000). Upland spate streams are very

235   characteristic of peat catchments in the west of Ireland, particularly within the

8

236   Burrishoole catchment (Allott et al., 2005). The average slope across the buffer zone is

237   5 % and this increases to 35 % within 10 m of the stream, while the slope down the

238   stream bed is approximately 2.5 %. The stream bed consists of boulders and gravel,

239   while mineral-rich peat is evident along the banks and slopes adjacent to the

240   watercourse. Blanket peat of varying depth down to 2 m covers the site, which

241   overlays a sand and gravel layer on top of the Cullydoo formation of Srahmore

242   quartzite and schist bedrock (McConnell and Gatley, 2006). This blanket peat is an *in-

243   situ* blanket mire with an average gravimetric water content of greater than 85 %, dry

244   bulk density of approximately 0.1 g cm^{-3} and a mineral content of approximately 3 %.

245   Bedrock does not protrude the surface of peat and the minimum peat depth is 0.3 m.

246   Closer to the stream, the mineral-rich peat is at a shallower depth of less than 1 m, has

247   an average gravimetric water content of 35 %, a dry bulk density of approximately 1 g

248   cm^{-3}, and a mineral content of approximately 95 %. During the course of the study, the

249   RBZ had a yearly average water table depth of 0.17 m, while the average water table

250   depth in the standing forest was 0.42 m.

251

252   The site was planted in 1966 with Sitka Spruce (*Picea sitchensis*) and Lodgepole Pine

253   (*Pinus contorta*). In May 2006, an area of 2.49 ha 30 m north, 50 m south and 300 m

254   along the stream was clearfelled to create the RBZ (Ryder et al., 2011) (Figure 1). This

255   is wider than the current buffer width recommendation of 10 – 25 m. In line with best

256   management practice (BMP), brash mats were used to prevent soil damage by the

257   heavy logging machinery. These mats were created by the harvester, which laid the

258   logging residues of branches and un-merchantable logs in front of the harvester in

259   continuous, slope-dependant strips on which it travelled as it felled the trees. These

260   were left *in situ* on completion of clearfelling. Typical forest practice would normally

261   be to windrow these brash mats into regular rows away from the watercourse when

262   preparing the site for replanting. The direction and position of the brash mats on the

263   southern side of the RBZ are shown in Figure 1. No rutting due to brash mat use was

264   noted on site.

265

266   In April 2007, one year after felling, the area was replanted with native broadleaved

267   tree species from Coillte nurseries, including *Ilex aquifolium* (holly), *Sorbus aucuparia*

268   (rowan), *Alnus glutinosa* (common alder), *Salix cinerea* (grey willow), *Betula pendula*

269   (common birch) and *Quercus robur* (oak pedunculate). These saplings were all

270   containerised and of varying height ranges: (1) 0.4 – 0.8 m (birch, rowan and willow)

271   (2) 0.3 – 0.5 m (oak and alder) (3) 0.1 – 0.2 m (holly). All saplings were 2 yr old,

272   except the birch, which was 3 yr old. No fertilizer was applied and the area was not

273   cultivated, but the saplings were pre-treated by dipping in Dimethoate (pyrethroid

274   insecticide) to protect them against the pine weevil (*Hylobus abietis*) (Ryder et al.,

275   2011). This planting was not intended to be productive commercial forestry (expected

276   survival rates of > 90 % after 4 yr), but aimed to examine which species of trees had

277   the potential to establish and survive in a hostile peatland environment. The perimeter

278   of the created buffer zone was then fenced off to protect it from grazing by sheep and

279   wild animals, not including deer, as a sufficiently high (exceeding 2.1 m) fence was

280   not installed.

281

282   **2.2.   Vegetation**

283

284   A detailed description of the location and composition of the sapling planting regime

285   in April 2007, post clearfelling, was conducted by Ryder et al. (2011) (Table 1).

286   Thirty-three plots in total, 20 on the southern side and 13 on the northern side of the

287   stream, were planted in a 2 x 2 m block pattern with a red stake placed in the centre of

288   the plot for identification (Figure 2). Nine trees per plot were planted, totalling 297

289   saplings of various tree species across the site (birch, alder, rowan, willow, holly and

290   oak). No planting took place within 5 m of the stream. In August 2011, as part of the

291   present study, a survey was carried out to determine the percentage survival and

292   increase in height of the surviving saplings. An increase in height was measured as the

293   percentage change from the average original height (obtained from Coillte Nurseries,

294   pers. comm.) to the measured height on site in August 2011. For example, no change

295   in height was denoted as a 0 % change, while an increase in height from 0.4 m to 1 m

296   (a change in height of 0.6 m) was given a 150 % increase.

297

298   **2.3.   Water Analysis**

299

300   Subsurface and surface water samples were collected throughout the site and upstream

301   and downstream of the buffer (Figure 1), mainly during high peak or storm events

302   from April 2010 to April 2011 (n=5 dates) to examine the movement and

303   concentration of nutrients in the peat and surface runoff. Sampling was focused on the

304   RBZ, but also included the adjacent mature standing forest to allow comparison with

305   the original condition of the buffer prior to clearfelling in 2006. All samples were

306   grouped under four specific locations: (1) 1 m from the stream (within the RBZ) (2)

307   under brash mats (within the RBZ) (3) under the vegetated areas (within the RBZ but

308   not under brash mats) and (4) the standing forest. The standing forest to the left of the

309   RBZ (in Figure 1) is at the same topographical location as the RBZ, has a similar slope

310   and peat depth, and has similar mineral-rich peat near the stream. The direction of

311   groundwater flow on site was perpendicular to the stream and the brash mats.

312   Therefore, any vegetated areas within the RBZ, which had no brash directly on them,

313   were still influenced by the decaying brash material.

314

315   Standpipes were installed on site for subsurface water quality measurement and their

316   locations are illustrated in Figure 1. Each sampling location comprised a cluster of 3

317   sampling tubes positioned at 20 cm, 50 cm, and 100 cm depths below the soil surface.

318   Each standpipe consisted of a qualpex pipe with an internal diameter of 1.1 cm. Holes

319   were drilled in the lower 10 cm of the pipe and this was covered with gauze. A steel

320   rod was inserted into the pipe for support, as it was hand-pushed into the peat. The top

321   of the standpipes were covered to prevent the ingress of rain water. Any water lodged

322   in the bottom of the standpipe was removed under suction the day before water

323   sampling and the standpipe was allowed to fill overnight. Once extracted from the

324   pipe, water samples were filtered on site using 0.45 µm filters.

325

326   All water samples were returned to the laboratory and tested the following day or

327   frozen for testing at a later date. The water quality parameters measured were: (1) DRP

328   (2) ammonium-N (NH_4-N) (3) nitrate-N (NO_3-N) and (4) total oxidised nitrogen

329   (TON; NO_3-N + nitrite-N (NO_2-N)). All water samples were tested in accordance with

330   standard methods (APHA, 1998) using a nutrient analyser (Konelab 20; Thermo

331   Clinical Labsystems, Finland). Nutrient data were log_{10} transformed and analysed with

332   ANOVA (analysis of variance) in Datadesk (Data Description Inc., USA), to ascertain

333   the main sources of variation. Date, depth of soil where the sample was taken and the

334   location of the sample site were included as explanatory variables.

335

336   Inverse distance weighted (IDW) analysis was carried out on the study area using

337   ArcGIS (Release Version 9.3, Environmental Systems Research Institute (ERSI),

338   California, USA) to show the concentrations of nutrients under the decaying brash

339   mats. Inverse distance weighted analysis is a geospatial analytical tool which

340   interpolates between sampling points, giving a greater weight to values closest to the

341   cell value being interpolated. A 'halo' effect on individual standpipes can be caused

342   where very high concentrations are in close proximity to lower concentrations, giving a

343   shorter distance for interpolation between the points.

344

345   **2.4.   Soil Analysis**

346

347   Water extractable phosphorus and desorption-adsorption isotherm testing were carried

348   out on samples of the soil from the RBZ and the adjacent mature standing forest. For

349   both tests, a series of sampling points were selected in three transects parallel to the

350   stream in the RBZ at the following locations: (1) 1 m from the stream (n=10) (2) under

351   the brash mat approximately 35 m from the stream (n=20), and (3) under a vegetated

352   area in-between brash mats approximately 45 m from the stream (n=20). Soil samples

353   (n=10) were also collected from the mature standing forest to represent the

354   contributing area. To select the sampling locations, a grid was laid out on the standing

355   forest, and soil samples were extracted at random locations on the grid. Samples were

13

356 extracted with a 30 mm-diameter gouge auger after clearing of the Oi horizon, litter

357 layer, which was mainly composed of degrading moss and needles. Väänänen et al.

358 (2007) found that this layer had the lowest P retention capacity and it was therefore

359 omitted from testing in the present study. Samples were then placed in sealed bags on

360 site and were homogenized by hand in the laboratory.

361

362 For WEP tests, samples were collected in spring 2011 at two depths, 0 – 0.1 m and 0.1

363 – 0.5 m, along each transect. Sub-samples of peat (n=5 from each depth), equivalent to

364 1 g dry weight, were mixed with 30 ml of deionized water and shaken for 30 min at

365 225 rpm using a rotary shaker after Rodgers et al. (2010). The filtered supernatant

366 water (filtered with 0.45 µm filters) was tested using a nutrient analyser (Konelab 20;

367 Thermo Clinical Labsystems, Finland). The remaining soil sample was used to

368 determine the gravimetric water contents.

369

370 For desorption-adsorption isotherm testing, the soil samples were collected at depth

371 increments of 0 – 0.15 m and 0.15 – 0.30 m below the soil surface. Phosphorus

372 solutions were made up to concentrations of 0, 0.2, 0.5, 1, 2, 3.5 and 10 mg P L^{-1}. Sub-

373 samples of peat (n=3 for each depth and each concentration), equivalent to 1 g dry

374 weight, were mixed with 40 ml of the P solution and shaken for 1 hr at 180 rpm using

375 a rotary shaker. The 10 mg P L^{-1} solution was only used for the samples collected 1 m

376 from the stream due to the high mineral content. The solutions were then allowed to

377 stand for 23 hr before being placed in the shaker again for 5 min at 120 rpm after

378 Väänänen et al. (2008). The filtered (0.45 µm) supernatant water was tested using a

379 nutrient analyser (Konelab 20; Thermo Clinical Labsystems, Finland). The remaining

380 soil sample was used to determine the gravimetric water contents and the mineral

381 content, which was determined by loss on ignition (LOI) at 550 °C (BSI, 1990).

382

383 **3. Results and Discussion**

384

385 **3.1. Vegetation**

386

387 The overall survival rate of the planted saplings by 2011 was relatively high, given that

388 the site was not cultivated, no fertiliser was applied and broadleaves were planted in an

389 environment hostile for their survival. Over half of oak (57 %), rowan (57 %) and

390 birch (51 %) saplings survived, suggesting that adequate nutrients were available in the

391 RBZ. However, the survival of willow (22 %), alder (25 %) and holly (44 %) was not

392 as successful, but evidence of surviving saplings was found in some plots. For saplings

393 of an original height and nature similar to that of this study, some degree of protection

394 and care should be afforded to ensure growth (M. Sheehy Skeffington, pers. comm.)

395 and vegetation control is vital for a plant to thrive (Renou-Wilson et al., 2008).

396 However, maintenance and grass removal is generally not performed on saplings

397 planted in peat areas. The study area was fenced off to protect the saplings from

398 grazing (except deer who could possibly jump the fence), but grass and other shading

399 species were not removed from around the saplings. *Sphagnum* spp., *Erica* spp. and

400 *Calluna* spp had naturally regenerated in the RBZ, and were likely to be competing

401 aggressively with the planted saplings for both nutrients and light. It is not known if

402 naturally regenerated ground vegetation alone would suffice for excessive nutrient

403 uptake if the native saplings were not planted in the RBZ. Further research should be

404   carried out to quantify the need for broadleaf plantation on clearfelled RBZs for

405   nutrient uptake.

406

407   Figure 3 shows the percentage increase in the height of the surviving saplings from the

408   various tree species. Even though 57 % of oak survived after 5 yr on site, there was

409   very little growth in the saplings (26 %). In comparison, the 51 % of birch that

410   survived had a percentage increase in height of 70 %. Similarly, Renou-Willson (2008)

411   found that oak seedlings only grew 13.9 cm in 3 yr where mounding was employed on

412   a cutaway peatland in the Irish midlands, whereas native birch was reported to grow up

413   to 50 cm per year in the same location (Renou et al., 2007). In the present study,

414   *Calluna vulgaris* was very evident on the site and was close to the oak plots; this may

415   have negatively impacted on the survival and growth of oak. Frost et al. (1997) noted

416   that competition from grass turf led to a significant increase in the seedling mortality

417   of oak species such as *Quercus robur* and observed that *C. vulgaris* had a negative

418   impact on the growth of seedlings of *Q. petraea* due to the hindering of the mycorrhiza

419   development (the mutually beneficial relationship between the fungus and the roots of

420   the plant). Rowan had the greatest increase in height over all other species, with a

421   growth of 73 % in surviving saplings. The willow that survived experienced no growth

422   over the study period (0 %) with the majority of the plants barely progressing beyond

423   seedling stage. Of the 12 alder plants that were originally planted, 3 survived, and only

424   one of these 3 surviving plants experienced growth during the 5 yr study period. Holly,

425   at approximately 0.15 m, was the smallest in size to be originally planted and the

426   surviving holly saplings (44 %) saw an overall growth of 22 %.

427

428 In the west of Ireland, natural broadleaf regeneration is rare and is largely limited to

429 river banks of higher mineral content than the surrounding infertile peatland

430 (Conaghan, 2007, unpublished report). No planting took place within 5 m of the

431 stream, but it was noted that there was a number of large trees (>2 m high) of various

432 species (oak, rowan and holly) on the banks of the stream, which were not planted

433 during the saplings' planting regime in 2007. Conaghan (2007, unpublished report)

434 observed that growth of broadleaved trees is more successful in previously forested

435 areas with a peat depth of less than 1 m, and also noted that sites with better drainage

436 and shelter fostered a better environment for the growth and survival of transplants.

437 The exposed nature of the Altaconey site and the depth of peat may have adversely

438 affected the growth rate of the willow, alder and holly species. Conaghan (2007,

439 unpublished report) also observed that the survival of willow cuttings placed directly

440 into the peat was low, but larger willow transplants, which were grown for a time prior

441 to transplanting in more fertile soil, did thrive. In the present study, the height of the

442 willow saplings planted in April 2007 was approximately 60 cm, but this did not

443 appear to survive (22 %) or grow (0 %) very well. Overall, the survival rate of the

444 planted saplings was relatively high, taking into consideration the lack of cultivation,

445 artificial fertilisation and maintenance, possibly grazing by deer and late planting of

446 the saplings. A greater growth rate may have been obtained if any of these

447 management techniques had been employed in the early stages of establishment.

448

449 **3.2. Water Analysis**

450

451 Date and depth were not significant sources of variation for any of the nutrient data,

452 while the location of the sampling site was significant for both DRP and NH_4-N

453 (ANOVA, $p<0.05$). Dissolved reactive phosphorus was significantly higher under

454 brash in the RBZ and was significantly lower close to the river ($p<0.05$, LSD post-hoc

455 test) (Figure 8). Ammonium-nitrogen in the standing forest was significantly higher

456 than that recorded under brash or in the vegetated parts of the RBZ ($p<0.05$, LSD post-

457 hoc test). Levels of TON were generally similar across all locations sampled (Figure

458 8). Surface and subsurface concentrations of NO_3-N were all low and were, in many

459 cases, below the limits of detection with a maximum concentration of 20 μg L^{-1}

460 (results not shown).

461

462 Inverse Distance Weighted images, generated from the subsurface DRP

463 concentrations, show the comparison of the RBZ with the standing forest (Figure 4).

464 This illustrates the higher nutrient concentration under the decaying brash mats in the

465 RBZ, which were left on site 5 yr before the present study. The DRP concentration

466 reduced close to the stream edge due to the adsorption capacity of the mineral-rich peat

467 near the stream (discussed in Section 4.3). The high concentrations of DRP in the

468 subsurface water under the brash mats and surrounding areas did not leach to the

469 stream, as frequent analysis of stream water upstream and downstream of the buffer

470 showed that it remained at between 4 – 10 μg L^{-1} (Figure 5). These values represent no

471 change from buffer creation (May 2006 - January 2007) (Ryder et al., unpublished

472 report), when MRP concentrations were approximately 5 μg L^{-1} in the stream. This is

473 similar to the concentration in the rain water in the area, which was approximately 6

474　μg L^{-1} (data not shown; figure based on 5 random rain samples analysed by the authors

475　over the study period).

476

477　Brash mats were created on site during clearfelling to protect the peat from

478　consolidation due to heavy machinery. Since creation of the RBZ in May 2006, 5 yr

479　prior to the present study, the peat was fertilised by these brash mats, as seen in Figure

480　4, and may easily have been removed from site for commercial purposes (Forest

481　Research, 2009). However, peat is considered highly vulnerable to a loss in fertility

482　and the removal of brash material from site can deplete the amount of base cations and

483　reduce available nutrients for the future growth of trees (Forest Research, 2009). The

484　potential loss of nutrients may be minimised through careful timing of brash removal

485　after the needles have fallen. Needles contain half-to-two thirds of total nutrients of the

486　brash material and the needle drop time period may occur anywhere from 3 to 9 mo

487　following clearfelling, depending on local climate and season (Forest Research, 2009).

488　Hyvönen et al. (2000) showed that after 6 – 8 yr of decomposition, needles and twigs

489　provided more nutrients for future tree growth as opposed to larger branches, and the

490　decomposition rate (and therefore nutrient addition) decreased with increasing branch

491　diameter. It was noted by Hyvönen et al. (2000), however, that 16 yr after clearfelling,

492　the decomposing branches increased the carbon content of the forest floor by up to 50

493　– 100 % and woody logging residues provided more N and P release than needles.

494　Nutrient release from decaying brash may enter sensitive receiving waters (Stevens et

495　al., 1995), but this did not occur at the site of the present study. However, if brash was

496　to be removed from peatland sites following fertilisation from needle drop time period,

497   other factors such as sediment release and economic value of degraded brash would

498   need to be considered.

499

500   **3.3.   Soil Analysis**

501

502   The WEP was highest in the vegetated areas and under the brash mats in the RBZ

503   (Figure 6) due to the leaching of P from the decaying brash mat into the soil in the 5 yr

504   since the creation of the buffer zone. Similar results were obtained by Rodgers et al.

505   (2010), who found that WEP under windrows of brash was significantly higher than

506   WEP in a windrow/brash-free area. The impact of the brash on WEP is a function of

507   the length of time it is left on site and the time taken for regeneration of vegetation to

508   occur (Macrea et al., 2005). The role of vegetation in nutrient uptake was investigated

509   by O' Driscoll et al. (2011), who measured WEP concentrations of 6–9 mg P kg^{-1} dry

510   soil from a seeded plot in contrast to 27 mg P kg^{-1} dry soil from an unplanted plot

511   positioned on a recently harvested site. As the brash mats run perpendicular to the flow

512   of water in the present study, any vegetated areas down-slope from the brash mats, but

513   not directly overlain by brash mats, were also affected. Water extractable phosphorus

514   decreased closer to the stream, but this reduction was due to the presence of a mineral-

515   rich peat along the bank of the stream, which protected the water-course from

516   increases in nutrient concentration. As there is a strong correlation between WEP

517   measured in peat and DRP concentration in surface runoff (O' Driscoll et al., 2011),

518   the potential for high concentrations of DRP in surface runoff from under the brash

519   mats is high.

520

521　The desorption-adsorption isotherms of P adsorption by weight (mg g^{-1} dry material)

522　showed that, while the P adsorption capacity was of the same magnitude in all areas

523　for the 0 – 0.15 m depth examined, more P adsorption took place at a distance of 1 m

524　from the stream at the 0.15 – 0.3 m depth (Figure 7). This was due to the mineral-rich

525　peat at this location. At both depths, the peat directly underneath the brash mat

526　appeared to be at P saturation and had little remaining adsorption capacity (reflecting

527　the high DRP concentrations at these points; Figure 4). As is typical for soils with a

528　low P retention capacity, the P adsorbed to the peat continued to rise with higher P

529　solutions and a maximum adsorption value was not reached (Väänänen et al., 2007).

530　Desorption of P from the peat in all areas and at both depths occurred when it was

531　overlain with water with a P concentration of 0 mg L^{-1} (Figure 7). This was greatest

532　under the brash mats and under the vegetated areas, especially at the 0 – 0.15 m depth.

533

534　When the results were expressed per volume of dry material (mg cm^{-3}) and the bulk

535　densities of the mineral-rich peat soil 1 m from the stream (approximately 1 g cm^{-3})

536　and the peat further up the buffer (35 – 45 m from the stream; approximately 0.1 g cm^{-3}

537　) were considered, the differences in the adsorption capacity of the soils was more

538　pronounced: the P adsorption capacity of the mineral-rich peat 1 m from the stream

539　were much higher than the peat layers further up the buffer. Loss on ignition analysis

540　showed that there was 90 – 95 % mineral content in the samples 1 m from the stream,

541　while the samples further back from the stream (35 – 45 m) had a 98 – 100 % organic

542　content. This trend was similar to what was found by Väänänen et al. (2006) in their

543　study on peat and mineral soils in Finland.

544

4. Conclusions

1. The created RBZ was capable of providing nutrients to planted saplings, fertilizing the peat with degrading brash material and preventing elevated levels of nutrients entering the adjacent water-course. This indicates that a created RBZ is a realistic management option in peatland forests.

2. The overall survival rate of the planted saplings in the RBZ was relatively high, with over half of oak, rowan and birch saplings surviving after 5 yr. The survival of willow, alder and holly was not as successful, possibly due to a number of factors including the exposed nature of the site, peat depth, maintenance, cultivation and fertilization.

3. Dissolved reactive phosphorus concentrations were significantly higher under the brash mats in the RBZ compared to all other areas. These high concentrations of DRP were due to the degrading brash mats left on site following clearfelling. However, this did not leach to the stream as the concentration of DRP upstream and downstream of the buffer remained low throughout the study. The standing forest had the highest concentrations of NH_4-N, while TON was similar for all areas. It is recommended to leave brash mats on site following clearfelling to fertilise the site and to reduce disturbance to the vulnerable peat sites.

4. Water extractable phosphorus and desorption-adsorption testing also confirmed the high concentrations of P under the brash mats. Water extractable phosphorus was highest in the vegetated areas and under the brash mats in the

568   RBZ. Desorption of P was highest under the brash mats and adsorption was

569   greatest in the mineral-rich peat soil adjacent to the stream.

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571   **5.   Acknowledgements**

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Figure 1. Location of Altaconey Riparian Buffer Zone (RBZ) with all standpipes (20,

741   50 and 100 cm depths), stream sampling locations upstream and downstream of buffer,

742   and rain gauge.
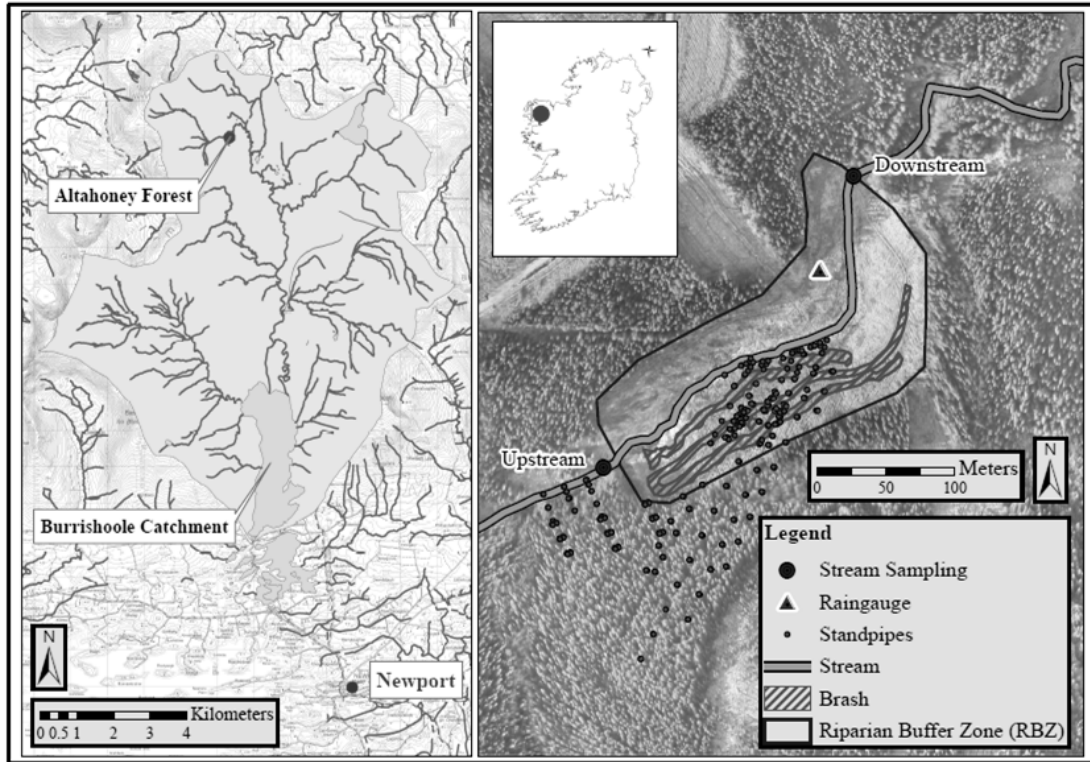
754   **Figure 2.** Sapling plot planting locations
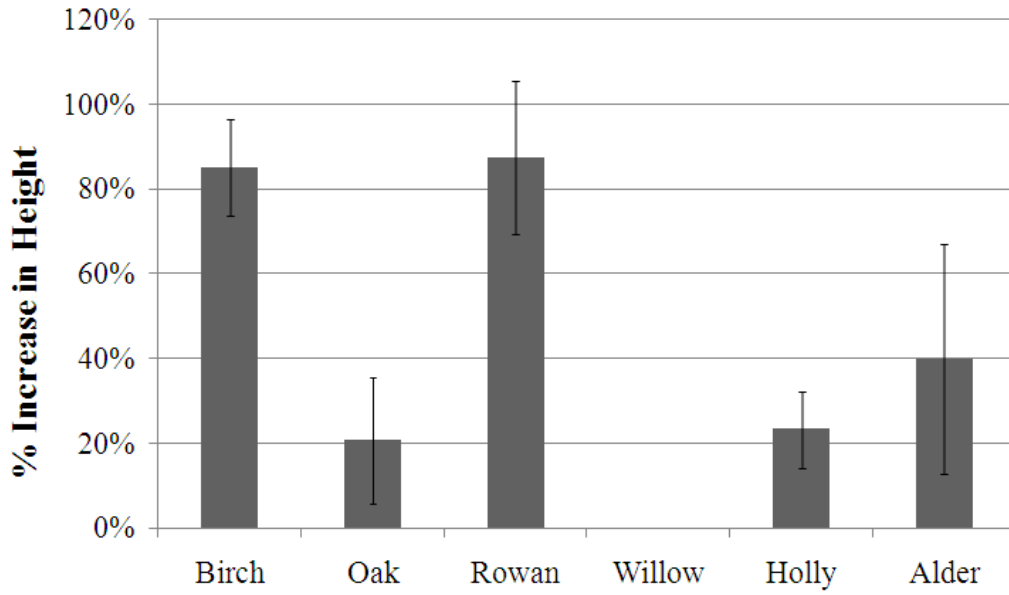

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768   **Figure 3.** Percentage average increase in height of surviving saplings on site from

769   April 2007 to August 2011 per tree species. Error bars indicate standard error.

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783   **Figure 4.** Average dissolved reactive phosphorus (DRP) concentration from 20 cm, 50

784   cm and 100 cm depths below the ground surface measured over a 12 month period

785   (April 2010 – April 2011) and expressed as μg L^{-1}
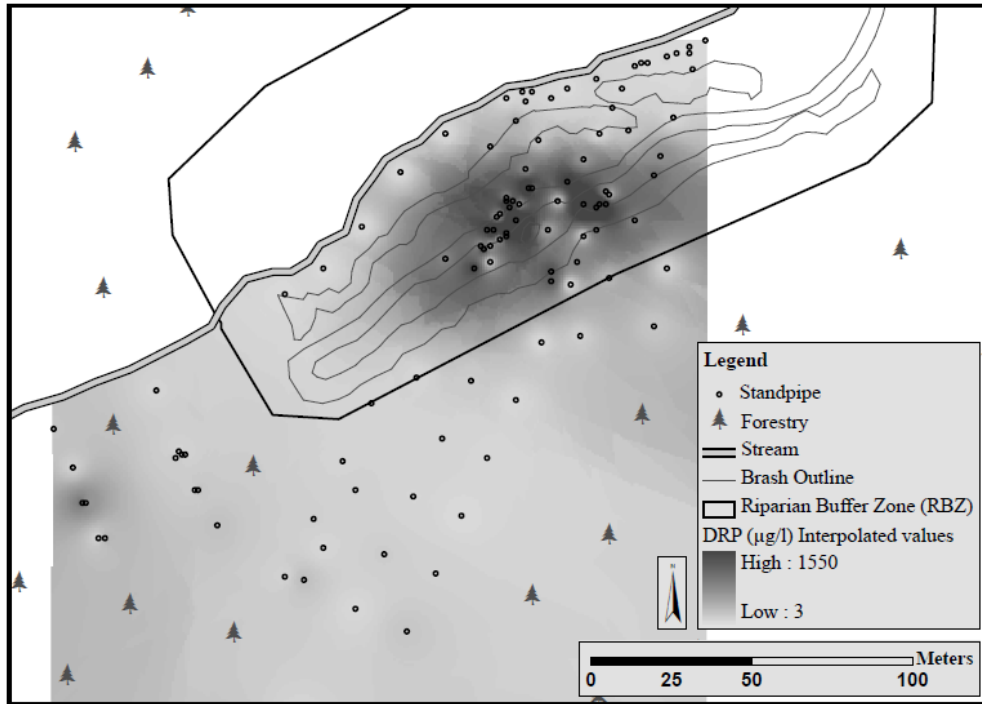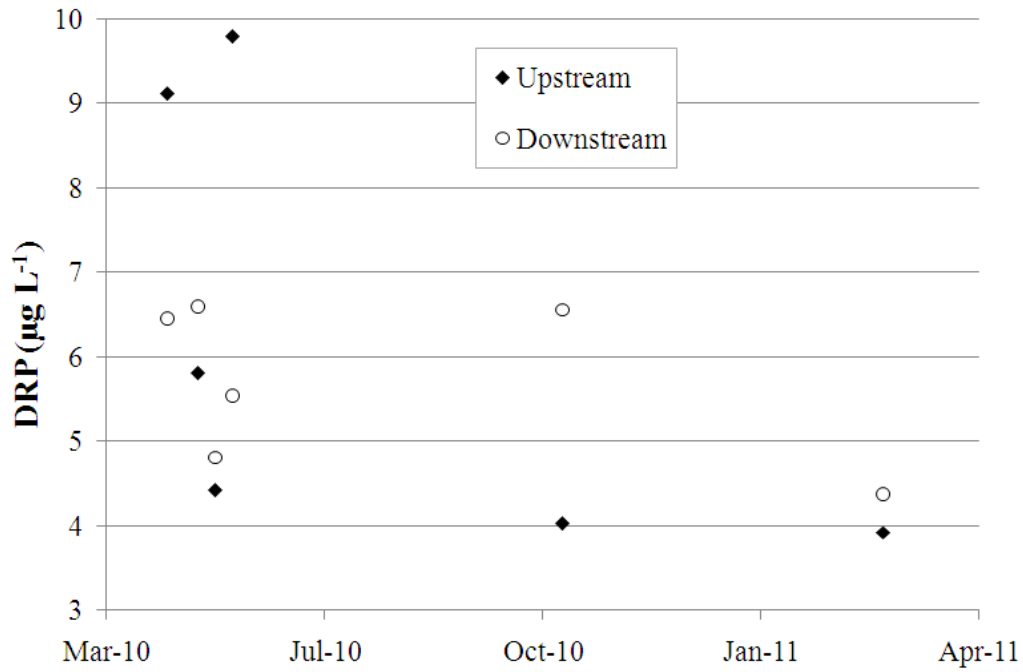
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797 **Figure 5.** Dissolved reactive phosphorus (DRP) concentration measured over a 12

798 month period (April 2010 – April 2011) and expressed as µg L^{-1} in stream water

799 upstream and downstream of the RBZ.

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813   **Figure 6.** Water extractable Phosphorus (WEP) concentration (mg kg^{-1} dry soil) in

814   riparian buffer zone (1 m from the stream, under brash and vegetated area) and forest

815   at 0 – 0.1 m and 0.1 – 0.5 m depths. Error bars indicate standard deviation.

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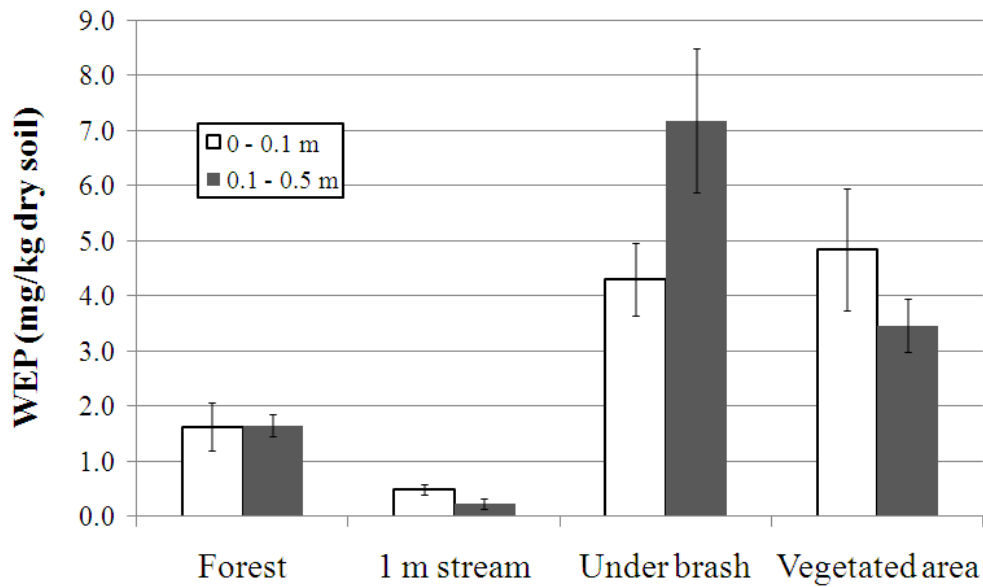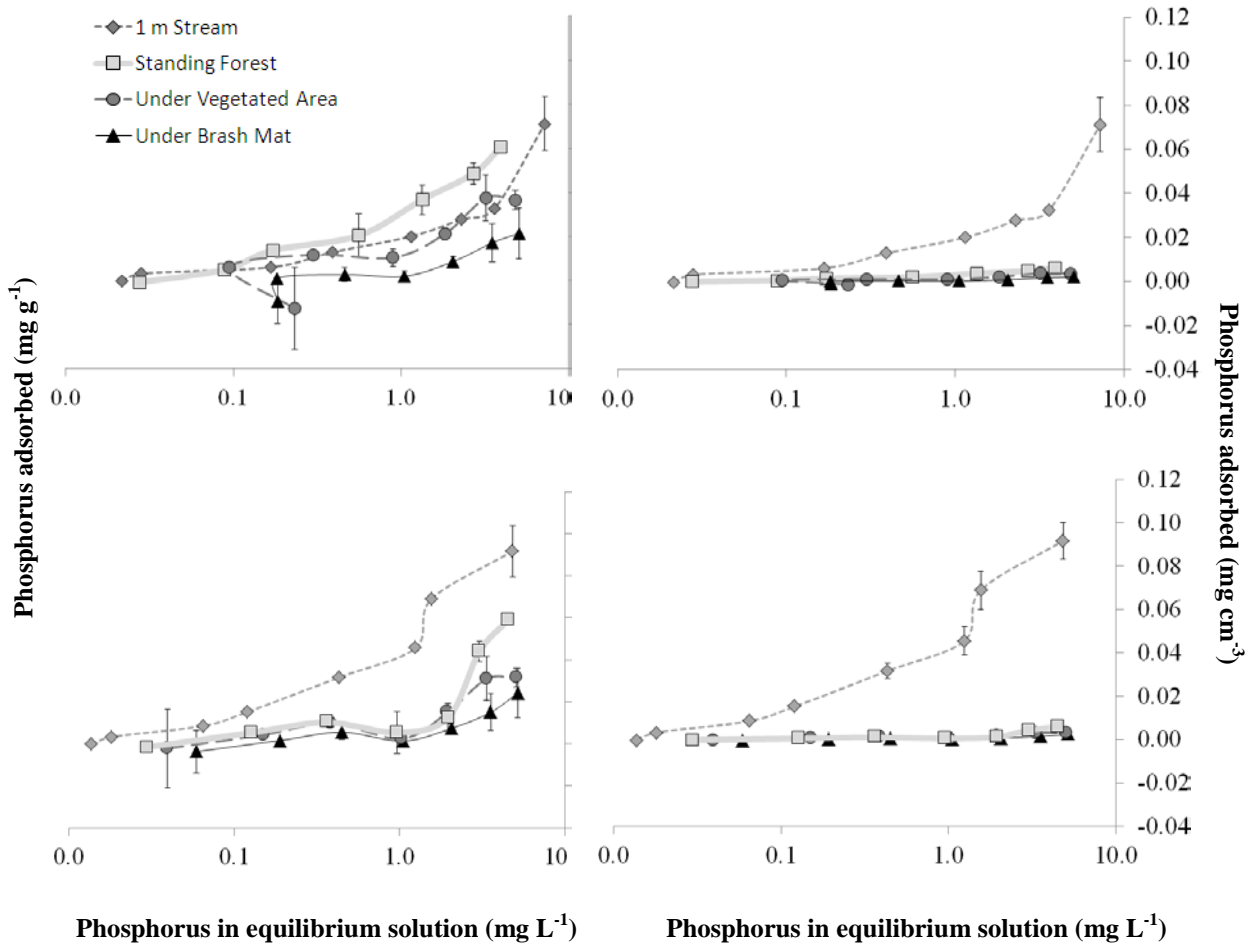

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Figure 7: Phosphorus (P) adsorption isotherms in riparian buffer zone and forest by

weight (mg g^{-1}) on left and by volume (mg cm^{-3}) on right at 0 – 0.15 m (top) and 0.15-

0.30 m (bottom) depths. Log scale on X axis for clarity.

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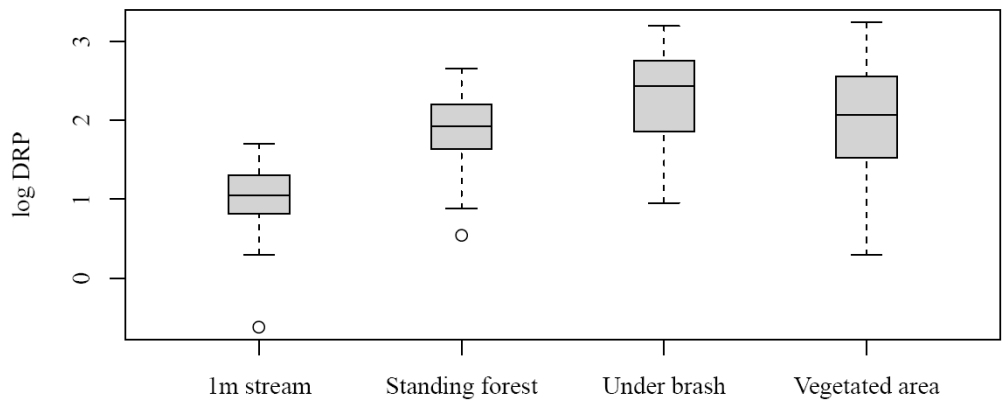838

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840 **Figure 8:** Box Plots of dissolved reactive phosphorus (DRP) (top), ammonium-N

841 (NH$_4$-N) (middle) and total oxidized nitrogen (TON) (bottom) for regenerated buffer

842 area (1 m from the stream, under brash mats and under the vegetated area) and

843 standing forest at 20 cm, 50 cm and 100 cm depths from April 2010 – April 2011. All

844 units are μg L^{-1}.

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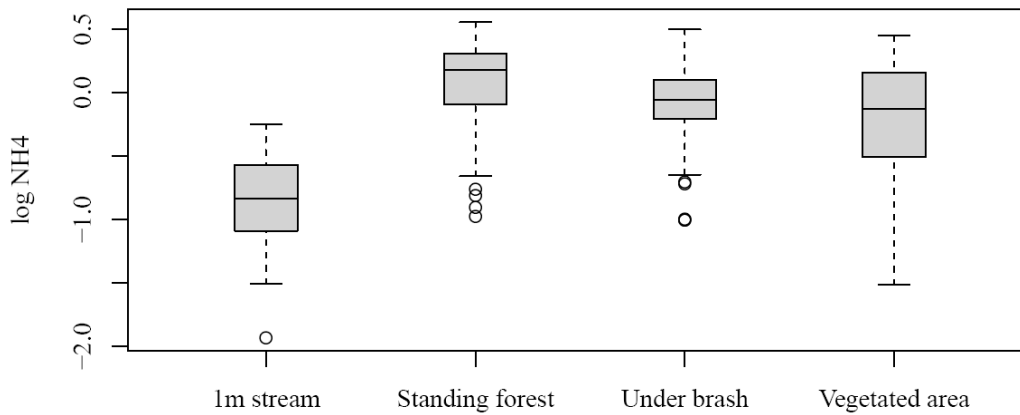
846
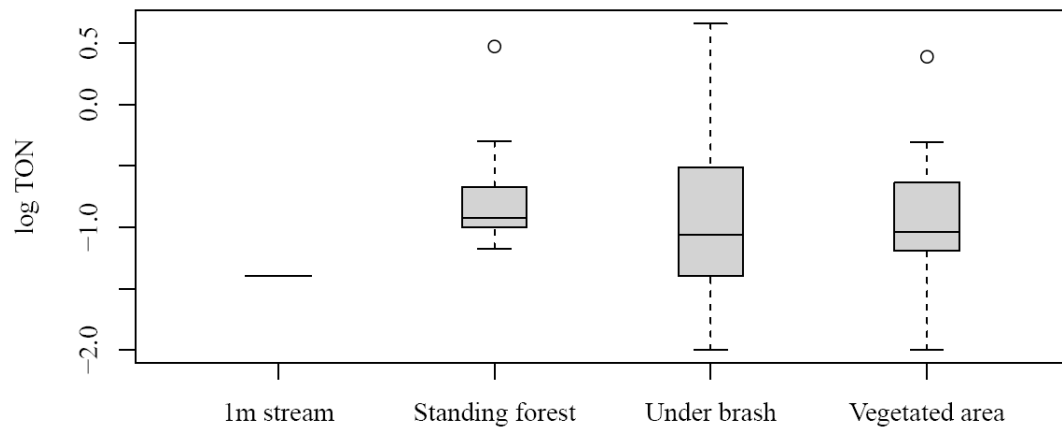
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Table 1: Description of the location and composition of the sapling planting regime in

April 2007, post clearfelling and surviving trees in August 2011.

Plot No.	April 2007						August 2011					
	Birch	Oak	Rowan	Willow	Holly	Alder	Birch	Oak	Rowan	Willow	Holly	Alder
1	9	-	-	-	-	-	7	-	-	-	-	-
2	-	9	-	-	-	-	-	8	-	-	-	-
3	-	9	-	-	-	-	-	5	-	-	-	-
4	9	-	-	-	-	-	6	-	-	-	-	-
5	-	-	9	-	-	-	-	-	3	-	-	-
6	-	9	-	-	-	-	-	0	-	-	-	-
7	-	9	-	-	-	-	-	8	-	-	-	-
8	9	-	-	-	-	-	2	-	-	-	-	-
9	-	9	-	-	-	-	-	2	-	-	-	-
10	-	9	-	-	-	-	-	4	-	-	-	-
11	9	-	-	-	-	-	9	-	-	-	-	-
12	-	9	-	-	-	-	-	9	-	-	-	-
13	6	-	3	-	-	-	3	-	1	-	-	-
14	9	-	-	-	-	-	3	-	-	-	-	-
15	6	-	3	-	-	-	6	-	1	-	-	-
16	6	-	3	-	-	-	2	-	2	-	-	-
17	-	-	9	-	-	-		-	4	-	-	-
18	9	-	-	-	-	-	9	-	-	-	-	-
19	6	-	3	-	-	-	4	-	2	-	-	-
20	6	-	3	-	-	-	4	-	3	-	-	-
21	9	-	-	-	-	-	3	-	-	-	-	-
22	6	-	-	-	-	3	3	-	-	-	-	1
23	6	-	-	-	-	3	1	-	-	-	-	1
24	-	-	-	9	-	-	-	-	-	2	-	-
25	9	-	-	-	-	-	6	-	-	-	-	-
26	-	-	9	-	-	-	-	-	2	-	-	-
27	6	-	-	-	-	3	2	-	-	-	-	1
28	6	-	-	-	-	3	2	-	-	-	-	-
29	6	-	3	-	-	-	-	-	1	-	-	-
30	-	-	6	-	3	-	-	-	5	-	1	-
31	-	-	6	-	3	-	-	-	6	-	2	-
32	-	-	6	-	3	-	-	-	6	-	1	-
33	9	-	-	-	-	-	0	-	-	-	-	-
Total	141	63	63	9	9	12	72	36	36	2	4	3
						% Survival	51%	57%	57%	22%	44%	25%