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5 HYBRID POPLAR PLANTATIONS IN A FLOODPLAIN HAVE BALANCED IMPACTS

6 ON FARMLAND AND WOODLAND BIRDS

7

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23 **Abstract**

24 Hybrid poplar plantations are increasing worldwide and are often accused of impoverishing
25 bird communities in surrounding farmland and woodland areas. We conducted 124 bird point
26 counts in a landscape where plantations, semi-natural forests and farmland occupied similar
27 surface areas. As expected, birds occurred at higher densities in forests than in plantations,
28 mostly due to the scarcity of late-successional forest birds in plantations. Contrary to
29 expectations, bird communities were the poorest in farmland dominated areas and the most
30 specialised in areas dominated by young plantations. Indeed, many grassland species,
31 including some of conservation concern in Europe, frequently or almost exclusively used
32 young plantations. However, plantations have probably depleted grassland bird communities
33 by fragmenting open areas, while playing a limited positive role on forest species. Total
34 length of unpaved roads had a positive effect on community specialisation index, while total
35 length of paved roads and mean forest/plantation patch size played no role. Bird density
36 increased with the development of the understory vegetation in mature poplar plantations. We
37 conclude that poplar plantations should be avoided in areas of high conservation value; in
38 other areas, they may increase the overall diversity of bird communities. At the plantation
39 scale, we advocate stopping understory removal as soon as possible and maintaining old trees
40 and hedgerows alongside or within plantations to provide suitable breeding habitats for bird
41 species that may be foraging in the plantations.

42

43 *Keywords:* Bird; Hybrid poplar; Landscape; Plantation; Management

44

45 **1. Introduction**

46 Many fertile floodplains in Europe were occupied by large riparian forests dominated
47 by hardwood tree species (oaks, ash) before the Middle Ages (Petts et al., 1989). Human
48 population growth caused a progressive conversion of a large part of these forests into
49 farmland, while most of the remnants were managed as coppices or coppices with standards
50 for fuel wood production (Steiger et al., 2005). The farming on these lands varied greatly over
51 time and space, and the landscape included pastures, hayfields, various ploughed fields, and
52 more recently maize fields and set-asides. From the XVIIIth century onwards, poplars were
53 often planted at the expense of arable lands and the few forest remnants (Petts et al., 1989).
54 Many factors jointly caused an acceleration of this phenomenon after the 1950s: the
55 embankment of large rivers limited the occurrence and magnitude of floods that poplar
56 plantations are sensitive to, the overproduction of dairy and meat production in Europe caused
57 a strong decrease in cattle numbers, the need for wood fibre increased, poplar plantations
58 (from two-year stems) readily took , and fast-growing, fungus-resistant cultivars became
59 available through hybridization among European, Asian and American *Populus* species
60 (Schnitzler, 1994). In France, hybrid poplar plantations now represent 1.6% of the forested
61 area (ca 260,000 ha) and this area is still increasing (+0.8% between 1988 and 2002,
62 SCEES/Terruti data 2003).

63 Since 1980, hybrid poplar plantations have been accused of contributing to the
64 decrease in bird populations in floodplains in Western Europe (Zollinger and Genoud, 1979;
65 Schmitz, 1986; Pont, 1987; Mourgaud, 1996; Godreau et al., 1999). However, this seems to
66 be an over-simplification. Indeed, several studies have shown that some uncommon/rare birds
67 may use poplar plantations; for example, the Golden oriole *Oriolus oriolus* is restricted to
68 mature poplar plantations in Great-Britain (Dagley, 1994). Furthermore, the ecological impact
69 of plantations depends on the habitats they are compared with: Hanowski et al. (1997) in

70 North America showed that the number of breeding birds was higher in poplar plantations
71 than in the row crops the plantations were replacing but less than in surrounding forest/shrub
72 habitat. Finally, extensively-managed plantations may be less unfavourable to breeding birds
73 than intensively-managed ones (Pont, 1987; Godreau, 1998).

74 We believe further studies are needed to confirm some of these results and better
75 understand the impact of hybrid poplar plantations on bird communities. In particular, the
76 previous studies rarely used a landscape perspective (but see Hanowski et al., 1997; Godreau,
77 1998) and yet, many birds need a mosaic of adjacent habitats to breed and forage (e.g.
78 Virkkala et al., 2004). Furthermore, the poplar plantations studied so far have usually been
79 isolated within a farmland matrix and have always represented a minor proportion of the
80 landscape (e.g. at most 12% in area, Godreau, 1998) and yet isolation of forest fragments is
81 well-known for causing deleterious effects on bird communities, irrespective of tree species
82 (Blake and Karr, 1987; Hinsley et al., 1995). More connected poplar plantations may host
83 more woodland species, as suggested by Godreau (1998).

84 In this study, we focused on a floodplain area in northern France where semi-natural
85 forests, poplar plantations and farmland represented equivalent proportions in the landscape.
86 Within this landscape, we assessed how bird communities responded to habitat composition at
87 three different spatial scales (100, 250 and 500m-radius circles). More specifically, our main
88 working hypotheses were:

- 89 - bird densities decrease with an increasing proportion of poplar plantations in the
90 surroundings,
- 91 - the greater the area planted with hybrid poplar, the less specialised the bird
92 communities,
- 93 - the presence of forest birds depends mainly on semi-natural forest areas but also, to a
94 smaller extent, on mature poplar plantations,

- 95 - bird densities vary with the fragmentation of habitats,
96 - bird densities in plantations increase with understory vegetation cover due to larger
97 areas suitable for breeding/foraging and/or to a higher number of potential niches.

98

99 **2. Methods**

100

101 *2.1. Study area*

102

103 The 6950-ha study area is located between Romilly-sur-Seine and Méry-sur-Seine
104 (France, Aube department, ca 100-120m asl), along 20 km of the Seine River on deep, chalky
105 soils corresponding to modern alluvial deposits. The climate is Atlantic with continental
106 influences (mean annual temperature ca 10°C with 630-700mm annual rainfall). Poplar
107 plantations represent 27% of the area due to the unusual width (2-3km) of the floodplain
108 (nearby confluence of the Seine with the Aube River). Other land uses found in the floodplain
109 were farmland and open areas (36%), forests (22% including abandoned poplar plantations),
110 urban areas (11%) and aquatic areas (4%). The poplar plantations did not differ from that of
111 forests in terms of median size (Wilcoxon rank-sum test, $W = 151758$, $P = 0.52$) but other
112 areas (farmland, other open, urban and aquatic areas) were significantly larger than poplar
113 plantations ($W = 226243$, $P = 0.001$) and forests ($W = 155897$, $P = 0.02$) (Table 1). The study
114 area extended slightly beyond the floodplain bottom onto the intensive row crop fields of the
115 plateau, so as to allow us to calculate habitat areas surrounding bird sampling points up to
116 500m. Crop fields are also found within the plain (including maize fields), along with set-
117 asides and, to a far lesser extent, hay pastures. The nearest large forest (the “Traconne”) state
118 forest that may be a source of forest birds is about 10km from the study area. As hedgerows
119 are also scarce on the plateau, bird exchanges between forested areas in the valley and the

120 plateau are probably limited. At a larger scale (Aube and Seine floodplains *sensu* the French
121 National Inventory: 34,025 ha), hardwood forests decreased by 53% between 1974 and 1994,
122 representing only 5.5% of the total area in 1994 (ca 1,660ha) *versus* 3,740ha for poplar
123 plantations (French Forest Inventory data, 1994).

124 The study area was first mapped from colour aerial photographs taken in 2001 and
125 2004 (NGI data). We separated unpaved and paved roads and identified nine major habitats:
126 (1) very young VYP, (2) young YP and (3) mature MP poplar plantations; (4) young YF and
127 (5) old OF hardwood forests; (6) farmland Farm (cropland and grassland combined); (7)
128 urban areas; (8) aquatic areas (rivers, ponds, lakes) and (9) other open habitats. We attributed
129 one of these habitat classes to the 1,592 polygons that were individualised from aerial photos;
130 then, in the field, we checked the correctness of this attribution. In the field, poplar stands
131 were attributed to the very young class when the mean diameter at breast height (dbh) was
132 less than 20cm, to the young class when the dbh was between 20 and 30cm, and to the mature
133 class when above 30cm (i.e. when the canopy closes). In the area, poplars are cut when about
134 20 years old. Mean distance between poplars in plantations is almost invariably 7m in the area
135 (occasionally 6 or 8m). Plantations are ploughed, and sometimes fertilised, at plantation; the
136 clones most often used are *Populus x interamericana* “Beaupré” and *P. x canadensis* “I214”.
137 The understory vegetation is mechanically and/or chemically removed during the first years
138 after planting. This control may be either continued more or less regularly (intensive
139 management) or ceased (extensive). In the case of intensive management, the vegetation
140 understory is limited to sedges and grasses (including Nettle *Urtica dioica*). In extensively
141 managed plantations, a shrubby layer develops with Blackthorn *Prunus spinosa*, Common
142 Elder *Sambucus nigra*, Willow *Salix sp*, and Blue Bramble *Rubus caesius*. All the young
143 forest stands (YF) actually originated from former poplar plantations cleared in December
144 1999 after the Lothar windstorm (ca 30% of the plantations in the study area were cleared)

145 and never replanted: these stands could be considered either as very extensive young poplar
146 plantations (due to the presence of naturally resprouting poplars) or young semi-natural
147 forests (due to the rapid recolonisation by ash and other shrubs). Old forest stands (OF)
148 included a variety of forest types, from most to least common: (1) coppiced Hazel *Corylus*
149 *avellana* with standards of Pedunculate Oak *Quercus robur* and Ash *Fraxinus excelsior*, (2)
150 abandoned over-mature poplar plantations overgrown by Ash, (3) spontaneous riparian forests
151 (including the previously mentioned species, as well as Alder *Alnus glutinosa*, Sycamore
152 Maple *Acer pseudoplatanus*, Smooth-leaved Elm *Ulmus minor* and various willow species
153 *Salix sp.* along river beds) and (4) old Ash/Hazel coppices.

154

155 2.2. Bird point counts

156

157 124 bird point counts were located throughout the study area, based on a quasi-
158 systematic grid design (ca 400m between points). We located the majority of the points along
159 paths for efficiency and discretion (a few were carried out along roads with limited traffic at
160 the time of the counts). Each point was visited twice (two 5-min counts), first between April
161 16 and 27 (between 6:30 and 9:43 am), and second between the May 30 and June 8, 2007
162 (between 5:42 and 9:32 am). All counts were carried out under good weather conditions (no
163 wind, no rain).

164 During the count, the observer (FA) used aerial photographs centred on the points to
165 localise all singing (the great majority) or moving/flying (a minority) birds. The fact that the
166 landscape was very fragmented helped to locate the birds. As for the territory mapping
167 method, the observer estimated whether the birds recorded during the second visit had already
168 been detected during the first visit (according to the spatial proximity among the contacts). A
169 total of 2,482 individual birds (62 species) were recorded and incorporated in the GIS. Total

170 cover for the 1-4m and 4-8m vegetation layers was visually estimated (and classified in 5%
171 increments above a minimum 5% total cover). The mean poplar dbh was measured in poplar
172 plantations immediately surrounding the points (but not in more distant plantations).

173

174 2.3. Data analyses

175

176 To test our working hypotheses, for each point we estimated the bird density, the mean
177 specialisation and preference for mature forests of the bird community. Only birds recorded
178 within 100m of the observer were kept (N=1138, 47 species). To adjust raw count data for
179 imperfect detection, we treated the two visits as two replicate counts and calculated the total
180 number of individuals A (observed abundance) and the number of individuals recorded at
181 only one visit (n_1). We calculated the Jackknife 1 estimator ($Jack1 = A + n_1/2$) and its
182 standard deviation ($SD = \frac{1}{2}\sqrt{3n_1}$) for each point as an estimate of the total number of birds
183 (all species) present within 100m. This estimator assumes heterogeneity of detection among
184 species (Burnham and Overton, 1979), a classical phenomenon in bird communities
185 (Boulinier et al., 1998). Density (number of pairs. ha^{-1}) was then estimated as $Jack1/\pi$ (the area
186 100m around the point, in ha). In models, we used the inverse of the variance of the density
187 estimate to weight observations: observations for which density was known with lesser
188 precision were logically given smaller weight.

189 We calculated the community specialisation index (CSI), a measure of the mean
190 specialisation of the bird community (Julliard et al., 2003), using data from the specialisation
191 index of 105 common birds by Devictor et al. (2007). The CSI index is known to correlate
192 well with the level of landscape fragmentation and disturbance (Devictor et al., 2007). The
193 higher the CSI, the more specialised the bird community. We followed the same rationale to
194 define the community preference for mature forests (CPMF). Muller (1985) provided the

195 mean age of the forest stands for 40 common European bird species where these species were
196 observed in French oak regular high forest stands; stand age varied from 10 years for the
197 early-successional Willow Warbler *Phylloscopus trochilus* to 147 years for the late-
198 successional Short-toed Treecreeper *Certhia brachydactyla*. CPMF was calculated as the
199 mean stand age over all species recorded within 100m. The greater the CPMF, the more
200 typical the bird community to mature forests. In analyses, we kept only points for which CSI
201 and CPMF indices could be calculated from at least five species and we weighted the
202 observations in models using the inverse of the variance of the index estimate (as for species
203 richness).

204 Using the GIS, we extracted the area covered by the nine pre-defined habitats within
205 100, 250 and 500m of the point. We also recorded the cumulated length of (1) unpaved roads
206 and (2) paved roads within the same radii. Finally, we calculated the mean patch size 100, 250
207 or 500m around the point (MPS) for plantations and forest patches. We modelled the impact
208 of these explanatory variables on the different community indices defined above using
209 spatially-explicit GAM models. A generalized additive model (GAM) is a generalized linear
210 model (GLM) in which spatial autocorrelation is modelled using a non-parametric, smooth
211 function of the geographical coordinates of the study points (using the gam function from R
212 library mgcv). To limit spurious effects of non-focal habitats (aquatic, gardens, urban, open
213 habitats other than farmland), we excluded from our analyses 20 points where these habitats
214 represented more than 5% of the area within 100m around the point. The final data set
215 contained 104 points for species richness, 85 for the CSI and 79 for the CPMF. To identify
216 the spatial scale that could explain the largest part of the total variance in the community
217 indices, we built models including all nine selected variables (farmed area, very young
218 plantation area, young plantation area, mature plantation area, young forest area, old forest
219 area, mean patch size of plantations and forests, cumulative length of paths, cumulative length

220 of roads) calculated either on 100, 250 or 500m around the point. Then, we compared the
221 relative merits of the three models with AIC (the lower the AIC value, the better). To further
222 investigate how the community indices varied among the six habitat area variables, we used
223 the model with the lowest AIC value in the preceding step of the analysis and we
224 progressively aggregated habitat area variables whose model coefficients were roughly
225 similar until the AIC of the resulting simpler model did not decrease any further: this step of
226 grouping habitat variables revealed which (groups of) habitats had similar and distinct
227 impacts on the bird communities. The significance of the MPS and length of paths and roads
228 variables was directly tested from an ANOVA (using the ANOVA function in R).

229 It was difficult to reliably assess the impact of the understory vegetation in plantations
230 on birds. Indeed, many plantations were small, leading to potentially confounding edge
231 effects. Furthermore, the observer rarely stood within the plantations during the counts, so
232 that only part of the larger plantations may have been effectively “sampled” (a positive aspect
233 of this sampling design is nonetheless that singing birds were less disturbed). We addressed
234 this issue by restricting our analyses to birds recorded within 100m of the observer, in mature
235 plantations (mean dbh over 30cm) covering at least 1ha within a 100m radius of the point (ca
236 >30% of the circle area). We further excluded all narrow plantations to limit edge effects,
237 finally retaining a list of 28 patches of mature plantations. We then estimated a density per
238 patch by dividing the number of birds recorded in a given patch (all species confounded,
239 corrected for imperfect detection using the Jackknife 1) by patch size. We built simple linear
240 models on unadjusted bird density with the 1-4m and 4-8m cover indices as explanatory
241 variables. Unadjusted bird density is given in Table 3 for the 20 most common bird species in
242 the six main habitats considered in this study.

243

244 **3. Results**

245

246 Mean bird density (Jackknife1) over the 104 point counts was estimated at ca 3.8 ± 1.5
247 pairs.ha⁻¹ (assuming a singing male corresponds to a breeding pair). Little spatial
248 autocorrelation was found as the effect of the spatial smoother was never significant. Bird
249 density was correlated to neither CSI nor CPMF, but the latter two variables were highly
250 negatively correlated (Pearson's $r = -0.55$, $t = -5.7$, $df = 77$, $P < 0.001$). For the three indices,
251 the model including explanatory variables defined locally (i.e. 100m around the point) had a
252 significantly lower AIC value than models including variables defined at larger scales (250 or
253 500m) (Table 2). Models explained a reasonable part of the variance for CPMF but a more
254 limited part for CSI and bird density, reflecting either measurement error or less determinism
255 in these variables (Table 2).

256 Bird density primarily decreased with farmland area and was highest in forests (Table
257 2). Interestingly, several habitats contributed to the CSI and CPMF but at different levels: CSI
258 increased primarily with the area of very young and young poplar plantations and secondarily
259 with the area of farmland. Similarly, CPMF was mostly related to the area of old forests but
260 the negative impact of the main other habitats gradually decreased with the structural
261 similarity of the habitat to old forest (this similarity increasing from (1) very young
262 plantations, (2) farmland and young plantations, (2) young forest to (3) mature plantations).

263 None of the three indices was significantly related either to the mean patch size of
264 plantations and woodlands, or to the cumulative length of roads. Only the mean level of
265 specialisation of the species increased with the length of footpaths.

266 Variation in bird density between plantations increased with the increase in the
267 understory cover. Nonetheless, log-transformed bird density (to limit heteroscedasticity)
268 increased more or less linearly with the understory cover up to 30% (Pearson's $r = 0.62$,

269 $t=3.88$, $df= 24$, $P<0.001$) and then seemed to stabilise (Fig. 1). On the contrary, bird density
270 was unrelated to the overstory cover (Pearson's $r = -0.22$, $t=-1.1$, $df= 26$, $P=0.17$, Fig. 2).

271

272 **4. Discussion**

273

274 Our results show that the impacts of poplar plantations on bird communities are more
275 complex than it is usually claimed: birds in poplar plantations did occur in lower densities
276 than in semi-natural forest stands (whatever stand age) but in higher densities than in farmed
277 areas. Similarly, mature plantations effectively lacked many of the forest birds typically found
278 in old forest stands; yet a truncated succession in bird communities occurred along the poplar
279 rotation, the composition of bird communities in mature plantations being somewhat
280 intermediate between young and old forest stands. Finally, contrary to what is generally
281 thought, the most specialised bird communities were found in the very young and young
282 poplar plantations, whereas mature plantations and forests hosted the least-specialised bird
283 communities.

284

285 *4.1. Plantations versus farmland*

286

287 Bird communities were also found to be poorer in intensively-managed farmland than
288 in short-rotation hybrid poplar plantations in North America (Hanowski et al., 1997). Bird
289 populations in our study valley are probably also suffering from the intensive management of
290 the farmland. Obviously, the set-asides and the too few and too artificial grasslands play a
291 reduced protective role for grassland birds in the area. In addition, plantations may have
292 exacerbated the decline in grassland species by fragmenting the grassland areas. Indeed,
293 Godreau et al. (1999) showed that many grassland species were very sensitive to the

294 fragmentation of open habitats along the Saône River, a few hundred km south of our study
295 site. The relatively large areas planted with poplars in our site probably explain why many of
296 the specialist grassland species that could normally breed in our floodplain were either never
297 recorded (Quail *Coturnix coturnix*, Black-tailed Godwit *Limosa limosa*, Curlew *Numenius*
298 *torquata*, Meadow Pipit *Anthus pratensis*, Whinchat *Saxicola rubetra* and Yellow Wagtail
299 *Motacilla flava*) or very rarely recorded (Lapwing *Vanellus vanellus*, Skylark *Alauda*
300 *arvensis*, Corn bunting *Miliaria calandra*, Pied wagtail *Motacilla alba* and finches *Carduelis*
301 *sp.*). Yet, some of these species were regularly observed in the immediate neighbourhood of
302 the study area (the Skylark, Corn Bunting and Yellow Wagtail on the purely agricultural
303 plateau, finches in urban areas), supporting the hypothesis that the valley is repulsive to many
304 grassland birds. Finally, plantations probably favoured the Carrion Crow *Corvus corone*
305 which appreciates the mosaic of interconnected farmland and woodland patches; this species
306 is known to heavily predate on the eggs of grassland ground nesters (Andren, 1992).

307 Although poplar plantations at the landscape scale may contribute to the near absence
308 of many grassland species, several ground- and shrub-nesters usually found in farmland along
309 hedgerows were commonly recorded in very young and young plantations. The same pattern
310 was found along the nearby Saône River (Godreau et al., 1999). Interestingly, Yellowhammer
311 *Emberiza citrinella* was recorded as frequently in very young poplar plantations as in
312 farmland. Other species found in very young and young plantations included Melodious
313 Warbler *Hippolais polyglotta*, Whitethroat *Sylvia communis*, Grasshopper Warbler *Locustella*
314 *naevia* and, more marginally, Red-backed Shrike *Lanius collurio*. These species are
315 considered to be habitat specialists (Devictor et al., 2007), a fact that probably explains why
316 the most specialised bird communities were found in areas dominated by very young and
317 young plantations.

318

319 4.2. Plantations versus woodland

320

321 The lower bird density in plantations compared to semi-natural forests has repeatedly
322 been found in Western Europe and North America (Schmitz, 1986; Hanowski et al., 1997;
323 Godreau, 1998; Twedt et al., 1999). This pattern is mostly due to the scarcity in plantations,
324 even mature, of forest species that prefer old trees with a dense foliage to forage or breed such
325 as the Woodpigeon *Columba palumbus*, Blue Tit *Cyanistes caeruleus*, Nuthatch *Sitta*
326 *europaea*, Short-toed Treecreeper and Chaffinch *Fringilla coelebs* (Table 3). Although
327 cavity-nesters were logically more frequent in forests than in plantations, mature poplar
328 plantations in our study were regularly used by the Great-spotted Woodpecker *Dendrocopos*
329 *major*, the Great Tit *Parus major* (breeding in abandoned cavities of the Great-spotted
330 Woodpecker) and the Willow Tit *Poecile montanus*. Dead poplars may be suitable to this
331 species that needs decaying softwood trees to dig its own cavities.

332 The young stages of poplar plantations hosted bird communities globally distinct from
333 that of the young forests: species like Stock Dove *Streptopelia turtur*, Winter Wren
334 *Troglodytes troglodytes*, Nightingale *Luscinia megarhynchos*, Melodious Warbler, Blackcap
335 *Sylvia atricapilla*, Garden Warbler *Sylvia borin* and Starling *Sturnus vulgaris* preferred young
336 forests to young plantations, probably in response to the higher structural and compositional
337 diversity of the shrub layer (the starling was probably also favoured by the presence of old
338 poplar snags in many young forest patches). Conversely, the Tree Pipit *Anthus trivialis* and
339 Grasshopper Warbler were more frequently found in young plantations: they respectively
340 forage on or close to the ground, in dense herbaceous vegetation; these two species rapidly
341 colonise plantations after planting and desert them a few years later, as the canopy closes.

342 More surprisingly, the level of specialised bird communities was lowest in the areas
343 dominated by forests (at a level similar to that of mature poplar plantations). Actually, as for

344 farmland, among the forest specialist species potentially present in floodplain forests
345 (Godreau et al., 1999), some were never recorded (Middle-spotted Woodpecker *Dendrocopos*
346 *medius*, Wood Warbler *Phylloscopus sibilatrix*) or very rarely (Lesser-spotter Woodpecker
347 *Dendrocopos minor*, Spotted Flycatcher *Muscicapa striata*, Hawfinch *Coccothraustes*
348 *coccothraustes*). In our study, the semi-natural forest fragments are probably too small and
349 isolated to allow these area-sensitive birds to maintain viable populations.

350 Several authors argued that the development of understory in extensively-managed
351 poplar plantations is favourable to small birds (Schmitz, 1986; Godreau, 1998). We did in fact
352 observe that bird density increased with the vegetation cover between 1 and 4m (though
353 possibly not linearly), but this was not the case above 4m. Our sample size was too small to
354 statistically assess whether the higher bird density in extensively-managed plantations was
355 primarily caused by a greater number of species or simply by a higher abundance of the same
356 set of species. In the study by Godreau (1998), the number of species more frequently
357 recorded in intensively-managed plantations did not differ statistically from the number of
358 species more frequently found in extensively-managed plantations (16 *versus* 19, binomial
359 test, $P=0.74$). Thus, extensive management of the plantations probably improves the overall
360 quality of the mature plantations more than it provides new niches benefiting new species; as
361 a result, it probably mainly favours the species already able to cope with the initial intensive
362 management of poplar plantations. This hypothesis may also hold in North America since
363 Christian et al. (1996) also conclude that (aspen) stand thinning has a weakly positive and
364 transitory effect on species abundances but none on the number of breeding species.

365

366 *4.3. Impact of habitat fragmentation*

367

368 Godreau (1998) noted that the number and abundance of forest birds in poplar
369 plantations increased with the size of the plantation. We found little convincing evidence of
370 any effect of habitat fragmentation, except for a positive impact on the length of unpaved
371 roads on the mean community specialisation. The vegetation along unpaved roads is likely to
372 provide food resources that are otherwise limited in fields for several specialist birds,
373 especially seed-eaters such as yellowhammer and stock dove. The fact that the mean size of
374 forest and plantation patches (MPS) moderately varied among points (MPS within 100-m
375 radius disks exceeded 2ha for only eight of the 104 points) probably explains why we failed to
376 find an effect on bird communities. Similarly, paved roads were present within 100m of the
377 points in only 14 points (by comparison, 87 points included paths within 100m).

378

379 *4.4. Implications for bird conservation in floodplains*

380

381 This study gives a balanced view of the impact of poplar plantations on common birds
382 in floodplains. On one hand, plantations do not provide refuge habitats for the most
383 specialised grassland and woodland birds. On the other hand, several woodland birds use
384 mature plantations (although often at lower densities than in semi-natural forests). Thus, the
385 conversion of intensive crop fields by plantations may be beneficial to some open-land and
386 woodland birds (at different ages of the plantations). On one hand, in areas of high
387 conservation value, they probably contribute to the loss of the most area-sensitive birds in
388 floodplains by causing the loss and fragmentation of grassland and woodland. On the other
389 hand, in already fragmented and intensively-managed farming areas such as our study area,
390 many grassland birds, including some that are currently declining all over Europe, use young
391 plantations to breed (although it remains to be demonstrated that young plantations are not
392 also demographic sinks).

393 At the scale of the plantation, extensive management has a positive impact on bird
394 density. This means that the removal of understory should stop as soon as the understory no
395 longer threatens the growth of poplars. In addition, plantations seem to lack breeding places
396 for many species, although plantations may represent valuable foraging habitats. For instance,
397 the presence of the great tit in mature plantations probably indicates that other cavity-nesters
398 would breed in plantations if sufficient cavities would be available: thus, the retention of large
399 old/decaying poplars, ashes or oaks within or along the plantation would probably favour
400 cavity-nesters. Those trees with a dense crown would also represent breeding sites safer than
401 poplars for species nesting in tree crown. Alternatively, nest-boxes may be installed in
402 plantations as suggested by Twedt and Henne-Kerr (2001). Similarly, hedgerows may be
403 installed between plantations or within large plantations to the benefit of shrub bird species.
404 The supplementary cost related to the retention of old trees and hedgerows may be partly
405 compensated by a better regulation of poplar defoliating insect populations since these
406 measures would be mostly beneficial to insectivorous birds.

407

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413

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471 Table 1
 472 Impact of landscape variables on bird assemblages (BD bird density, CPMF community
 473 preference for mature forests, CSI community specialisation index)¹. a) AIC values of the
 474 three alternative models based on the same set of variables but defined at three different
 475 spatial scale (100, 250 and 500m around the point)); in bold the model with the lower AIC
 476 before aggregating habitat area variables (Farm to OF variables); the % deviance of this
 477 model in provided in b); in superscript AIC of the best model after aggregating the habitat
 478 area variables. c) Fitted coefficients and standard errors of the explanatory variables
 479 corresponding to the model in bold in a) (removing the intercept) and in superscript (a-d)
 480 variables whose aggregation improves model AIC (the AIC of the corresponding model is the
 481 value in superscript in a)). For MPS, Paths, Roads and the smoother: *P<0.05, italics P<0.1.

	Variables	BD	CPMF	CSI
a)	AIC (100/250/500)	617 ⁶¹² /619/632	587 ⁵⁸⁴ /599/610	-108 ⁻¹¹³ /-85/-82
b)	%Dev	36.8	67.9	39.1
c)	Farm	1.2±0.6 ^a	10.5±2.5 ^a	-0.27±0.02 ^a
	VYP	3±0.7 ^b	6.5±1.2 ^b	-0.20±0.03 ^b
	YP	3.1±0.7 ^b	11.2±2.1 ^a	-0.24±0.03 ^b
	MP	3.6±0.5 ^b	15.2±1.3 ^c	-0.33±0.02 ^c
	YF	4.5±1 ^c	12.4±1.9 ^a	-0.34±0.03 ^c
	OF	5.1±0.7 ^c	22.1±1.8 ^d	-0.34±0.02 ^c
	MPS	-1.0±0.8	<i>-4.5±2.3</i>	0.03±0.02
	Paths	0.3±0.5	-1.5±1.2	0.03±0.01*
	Roads	0.6±0.4	<i>1.7±0.9</i>	0.0±0.01
	Smooth df	<i>10.8</i>	5.8	3.2

482 ¹Farm farmland, VYP/YP/MP very young/young/mature poplar plantations, YF/OF young/old forests, MPS
483 mean patch size of plantations and forests (all in ha), Paths/Roads cumulated length of paths or roads (km);.
484 Smooth df: estimated degree of freedom of the spatial smoother.

485 Table 2

486 Bird density (number of singers/10ha) of the 20 most common species calculated from the

487 124 point counts of the study depending on the habitat where the individual was recorded

488 (#Ind: total number of individuals recorded within 100m around the points, see also Table 2

489 legend).

Species	#Ind	Farm	VYP	YP	MP	YF	OF
<i>Columba palumbus</i>	39	0	0	0.3	1.1	0.9	2.4
<i>Streptopelia turtur</i>	22	0.2	0	0	0.6	1.8	0.9
<i>Dendrocopos major</i>	14	0	0	0.6	0.8	0	0.3
<i>Anthus trivialis</i>	27	0.4	2.2	0.9	0.2	1.2	0.5
<i>Troglodytes troglodytes</i>	73	0	0.6	0.9	2.7	1.5	3.4
<i>Erithacus rubecula</i>	132	0	0.6	1.1	4.8	3.1	6.5
<i>Luscinia megarhynchos</i>	42	0	0.4	2.0	1.2	2.5	1.2
<i>Turdus merula</i>	50	0	0.4	1.4	1.5	1.8	2.2
<i>Turdus philomelos</i>	18	0	0.2	0	0.6	0	1.1
<i>Locustella naevia</i>	31	0.4	2.8	1.4	0.4	1.5	0
<i>Hippolais polyglotta</i>	17	0.2	0.8	0.3	0.4	1.5	0.2
<i>Sylvia communis</i>	46	0.4	3.2	1.4	1.0	2.8	0.3
<i>Sylvia borin</i>	27	0	0.2	0.9	1.0	1.5	0.8
<i>Sylvia atricapilla</i>	190	0.4	1.4	5.7	7.1	7.1	6.2
<i>Phylloscopus collybita</i>	117	0	2.0	4.0	5.1	2.8	2.8
<i>Cyanistes caeruleus</i>	44	0	0	0.9	1.1	1.5	2.6
<i>Parus major</i>	49	0	0	0.6	2.1	0.3	2.3
<i>Certhia brachydactyla</i>	19	0	0	0	0.4	0.6	1.3
<i>Sturnus vulgaris</i>	16	0.4	0	0.3	0.4	0.9	0.5

Chaffinch 77 0 0.2 1.1 1.9 1.5 4.9

490