



**HAL**  
open science

## Seasonal and spatial variations of stream insect emergence in an intensive agricultural landscape

Julien Raitif, Manuel Plantegenest, Océane Agator, C. Piscart, Jean-Marc Roussel

► **To cite this version:**

Julien Raitif, Manuel Plantegenest, Océane Agator, C. Piscart, Jean-Marc Roussel. Seasonal and spatial variations of stream insect emergence in an intensive agricultural landscape. *Science of the Total Environment*, 2018, 644, pp.594 - 601. 10.1016/j.scitotenv.2018.07.021 . hal-01848468

**HAL Id: hal-01848468**

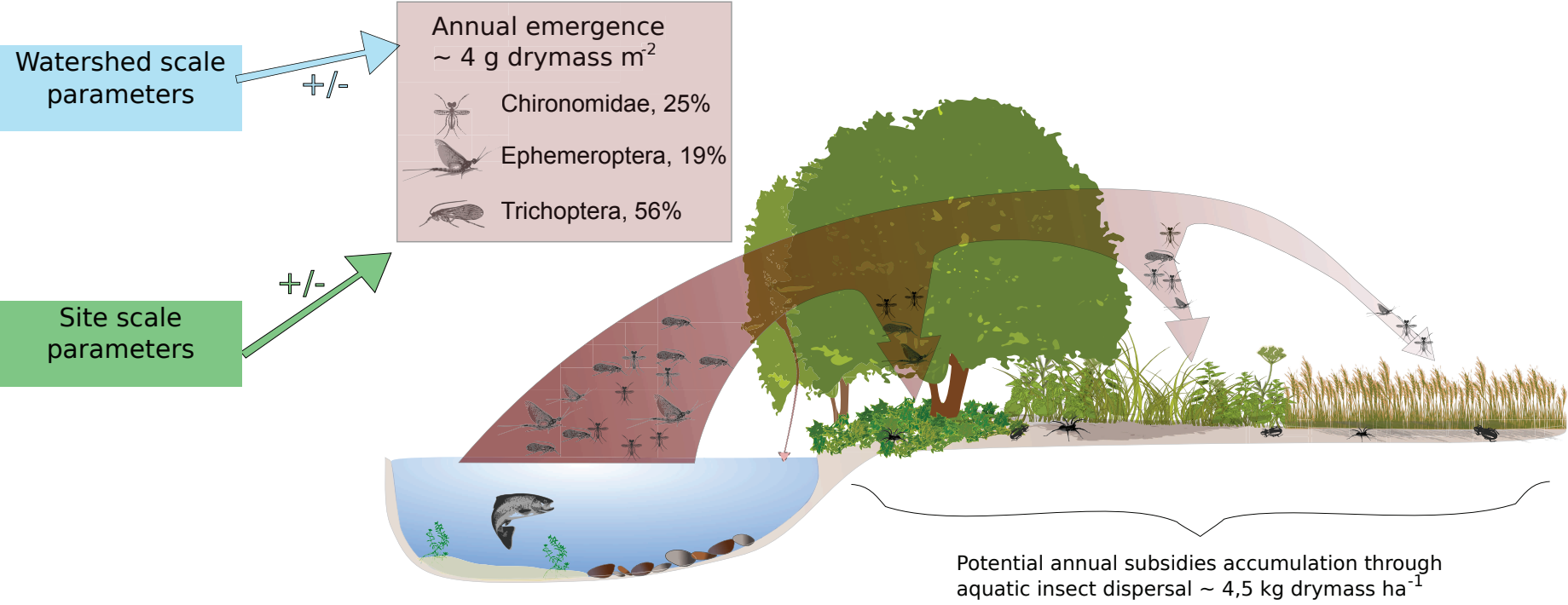
<https://institut-agro-rennes-angers.hal.science/hal-01848468v1>

Submitted on 11 Sep 2018

**HAL** is a multi-disciplinary open access archive for the deposit and dissemination of scientific research documents, whether they are published or not. The documents may come from teaching and research institutions in France or abroad, or from public or private research centers.

L'archive ouverte pluridisciplinaire **HAL**, est destinée au dépôt et à la diffusion de documents scientifiques de niveau recherche, publiés ou non, émanant des établissements d'enseignement et de recherche français ou étrangers, des laboratoires publics ou privés.

# AGROECOSYSTEMS



## Highlights

- Aquatic insects emergence reaches 4 g drymass (DM).m<sup>-2</sup>.y<sup>-1</sup> in agricultural streams
- Trichoptera contributed the most, followed by Chironomidae then Ephemeroptera
- Emergence happened throughout the year with taxon-specific patterns
- Several parameters linked to agriculture influenced emerging DM of aquatic insects
- We estimated potential deposit of aquatic subsidies on land at 4.5 kg DM.ha<sup>-1</sup>.y<sup>-1</sup>

# Seasonal and spatial variations of stream insect emergence in an intensive agricultural landscape

---

Julien Raitif<sup>1,2</sup>, Manuel Plantegenest<sup>1</sup>, Océane Agator<sup>2</sup>, C. Piscart<sup>3</sup>, Jean-Marc Roussel<sup>2,4</sup>

<sup>1</sup> IGEPP, Institut de Génétique, Environnement et Protection des Plantes, Agrocampus Ouest, 65 rue de Saint Briec, F-35042 Rennes, France

<sup>2</sup> INRA, Agrocampus-Ouest, Ecology and Ecosystem Health, 65 rue de Saint-Briec, F-35042 Rennes, France

<sup>3</sup> Université de Rennes 1 – UMR CNRS ECOBIO 6553 – Campus Beaulieu, 263 Av. du Général Leclerc, F-35042 Rennes Cedex, France

<sup>4</sup> Management of aquatic ecosystems in agricultural watersheds, INRA, Agence Française pour la Biodiversité, Rennes, France



1 *Abstract*

2 A growing amount of literature exists on reciprocal fluxes of matter and energy between  
3 ecosystems. Aquatic subsidies of winged aquatic insects can affect terrestrial ecosystems  
4 significantly, but this issue is rarely addressed in agroecosystems. By altering the production  
5 of benthic macroinvertebrates, agricultural practices could increase or decrease the strength of  
6 aquatic subsidies and subsequently the provision of several ecosystem services to agriculture.  
7 Effects of seasons and environmental variables on aquatic insect emergence were investigated  
8 in third-order agricultural streams in northwestern France. Most emerging dry mass (DM) of  
9 caught insects belonged to Trichoptera (56%), Chironomidae (25%) and Ephemeroptera  
10 (19%). We estimated that annual emerging dry mass of aquatic insects ranged between 1,445  
11 and 7,374 mg/m<sup>2</sup>/y depending on the stream. Seasonal variations were taxon-specific, with  
12 Ephemeroptera emerging only in spring, Trichoptera emerging in spring and early summer,  
13 and Chironomidae emerging throughout the year. The percentage of watershed area covered  
14 by agriculture, ammonium concentration and hypoxia positively influenced emerging DM of  
15 Chironomidae but negatively influenced Ephemeroptera. Emerging DM of Trichoptera and  
16 Ephemeroptera increased significantly as water conductivity and temperature increased.  
17 Channel openness increased the emerging DM of all taxonomic groups, but Chironomidae  
18 were more abundant in narrow, incised streams. Assuming that the biomass of aquatic  
19 invertebrates ultimately disperse toward terrestrial habitats, nutrient accumulations on land  
20 near streams were estimated to reach 0.5-2.3 kg C ha<sup>-1</sup> y<sup>-1</sup>, 0.1-0.5 kg N ha<sup>-1</sup> y<sup>-1</sup> and 0.005-  
21 0.03 kg P ha<sup>-1</sup> y<sup>-1</sup>, depending on the stream. This suggests a significant flux of aquatic  
22 nutrients to agroecosystems and the need for future studies of its potential influence on the  
23 ecosystem services provided to agriculture.

24

25 Keywords: agroecosystems; winged stream insects; emergence; aquatic subsidies;  
26 environmental factors; seasonal variations

## 27 1 Introduction

28 Riverine ecosystems host a rich but endangered biodiversity (Naiman and and Décamps 1997;  
29 Dudgeon *et al.* 2006) that could play a significant role in providing ecosystem services. In a  
30 recent review, Hanna *et al.* (2017) listed more than 30 types of services that riverine  
31 ecosystems provide, and observed a potential distinction between the location and the spatial  
32 extent at which these services are produced and delivered. Growing evidence indicates that  
33 ecosystems that were previously considered unrelated can interact. Several studies have  
34 demonstrated the existence of reciprocal exchanges of matter and energy between aquatic and  
35 terrestrial habitats (e.g. Polis *et al.* 1997, Richardson *et al.* 2010, Bartels *et al.* 2012). These  
36 “aquatic subsidies” can significantly influence the functioning of terrestrial ecosystems  
37 (Jackson and Fisher 1986; Havik *et al.* 2014; Dreyer *et al.* 2015), but this is poorly  
38 documented for agricultural landscapes. Aquatic subsidies can also influence several  
39 ecosystem services supplied to agriculture (Raitif *et al.*, in revision), and future studies are  
40 required to estimate the magnitude of their influence.

41 Adult aquatic winged insects are effective vectors of aquatic subsidies (Bartels *et al.* 2012)  
42 since they are abundant in almost all freshwater ecosystems, and their ability to disperse is  
43 substantial (Muehlbauer *et al.* 2014). Several studies have demonstrated their role as prey for  
44 terrestrial predators (e.g. birds, Gray 1993; carabids, Hering and Plachter 1997; spiders,  
45 Paetzold *et al.* 2005), providing a valuable source of nutrients for entire terrestrial ecosystems  
46 (Dreyer *et al.* 2015). Agricultural practices can regulate the production of benthic  
47 macroinvertebrates in streams by modifying water chemistry, eutrophication processes and  
48 pollution (Sallenave and Day 1991; Liess and Von Der Ohe 2005; Cross *et al.* 2006; Davis *et*  
49 *al.* 2010; Johnson *et al.* 2013a; Beketov *et al.* 2013); changing the flow regime and aquatic  
50 habitats (Rabeni *et al.* 2005; Kennedy and Turner 2011; Wagenhoff *et al.* 2011; Magbanua *et*  
51 *al.* 2016); and modifying riparian vegetation (Deegan and Ganf 2008) and water temperature

52 (Nagasaka and Nakamura 1999; Sponseller *et al.* 2001). Consequently, the influence of  
53 agriculture in watersheds on the emergence of stream winged insects could increase or  
54 decrease the strength of aquatic subsidies provided to terrestrial ecosystems. For instance,  
55 stream insect communities are usually dominated by small taxa with greater ability to disperse  
56 by flight in agricultural landscapes (Stenroth *et al.* 2015; Carlson *et al.* 2016; Greenwood and  
57 Booker 2016). While insect emergence depends on season and weather conditions (Corbet  
58 1964), few studies quantify these influences or analyze their variability in agricultural  
59 landscapes. For instance, Shieh *et al.* (2003) and Gücker *et al.* (2011) published seasonal  
60 variations of aquatic insect production in two agricultural streams, but not emergence data.  
61 However, information on spatial and temporal variation in the emerging biomass of aquatic  
62 insect taxa is essential to accurately estimate the amount of aquatic subsidies annually  
63 produced by agricultural streams.

64 In this study, we quantified the spatial and seasonal emergence of aquatic insects in 12 sites  
65 located in intensive agricultural landscapes. We aimed to (i) assess temporal variation in dry  
66 mass (DM) for the main emerging aquatic insect taxa and (ii) highlight the influence of  
67 environmental parameters at different spatial scales on emergence patterns. We hypothesized  
68 that agriculture intensification would drive emergence of aquatic insects at both local and  
69 watershed scales, and notably enhance Chironomidae (Diptera) that could emerge throughout  
70 the year.

## 71 2 Methods

### 72 2.1 Study sites

73 The study was performed in the Ille-et-Vilaine county (Brittany), western France. This area  
74 sits on sedimentary rocks (schist and sandstone) with occasional layers of aeolian loam  
75 deposits. The climate is oceanic, with a mean annual temperature of 10.5-12.5°C and

76 cumulative annual rainfall of approximately 700 mm (peaking in fall and winter) (Météo  
77 France 2017). Mean annual discharge of third-order streams in this part of Brittany is  $0.65 \text{ m}^3$   
78  $\text{s}^{-1}$ , ranging from  $0.06 \text{ m}^3 \text{ s}^{-1}$  in September (end of summer) to  $2.23 \text{ m}^3 \text{ s}^{-1}$  in March (end of  
79 winter).

80 Intensive agricultural practices that rely on large amounts of inputs (fertilizers and pesticides)  
81 and extreme modifications of the landscape have altered terrestrial and aquatic ecosystems for  
82 decades (Piscart *et al.* 2009). Stream banks, deeply incised into thick arable ground, are  
83 destabilized by clearcutting of riparian vegetation, which results in sloughing, bank erosion,  
84 siltation and homogenization of stream substratum. Twelve sites were selected from 8  
85 watersheds: Champagne, Roncelinçais, Jardière, Rocher, Tertre, Ourmais, Bray, Ormal,  
86 Moulin, Fèvre, Vallée and Sauvagère (Table S1). Selection criteria was based according to (1)  
87 third-order permanent streams (Strahler classification), and (2) approximately 6 m wide and  
88 flowing along small-grain cereal fields (winter wheat or barley). We chose third-order streams  
89 to avoid summer drying of lower-order streams and fall and winter floods in higher-order  
90 streams, which would have impeded aquatic insect sampling. A drought occurred in 2016,  
91 resulting in very low discharge in winter 2016/2017 ( $1 \text{ m}^3 \text{ s}^{-1}$ ). Mandatory grass strips (~8 m  
92 wide) separated streams from fields. This highly enriched buffer zone was dominated by  
93 nitrophilous plants belonging to a variety of families (Poaceae, Urticaceae, Apiaceae, *etc.*).  
94 Riparian vegetation near the stream bank consisted of shrubs, small trees (*Salix* sp.), and  
95 occasional larger trees (*Quercus* sp., *Alnus* sp.).

## 96 2.2 Aquatic insect sampling

97 We estimated aquatic insect emergence by deploying two emergence traps at each site. The  
98 traps consisted of a floating pyramidal tent ( $1 \text{ m}^2$  at its base) made of nylon mesh and  
99 anchored to shorelines with ropes. Because substratum and water velocity strongly affect  
100 aquatic insect communities (Tachet *et al.* 2010), one trap was set in a deep and silty habitat

101 and the other in a shallow gravel-pebble habitat to reflect the natural habitat heterogeneity of  
102 each site. Adult insects were collected in a plastic bottle placed at the top of the trap and filled  
103 with a mix of water, concentrated detergent and propylene glycol (approx. 20%) to preserve  
104 insects. Upon collection, aquatic insects were stored in alcohol (70%).

105 Six campaigns of sampling were performed in 2016 and 2017: 17 - 26 May; 6 - 15 June; 27  
106 June – 6 July; 12 - 21 September; 28 November - 5 December; 27 February – 9 March. At  
107 each site, emergence trap was collected after 7 consecutive days and nights. Ten days per  
108 campaign were necessary to proceed the 12 sites because of travel time between sites. The  
109 timeline was designed to match aquatic insect emergence in such streams, which occurs  
110 mainly in spring (March to June), to a lesser extent in summer and early fall (July to October),  
111 and almost stops during cold and high-discharge winter months (Corbet 1964). Sampling was  
112 carried out in December 2017 to confirm that insects do not emerge in coldest times at the end  
113 of fall or during winter, but only at one site, to avoid damaging the equipment in windy and  
114 flooded conditions. A total of 111 samples, totaling 777 days of aquatic insect emergence,  
115 were collected; 11 samples were lost due to strong winds, flash floods or vandalism.

### 116 2.3 Emergence abundance, dry mass and secondary production

117 A stereomicroscope was used (1) to identify adult aquatic insects to the order level for  
118 Ephemeroptera, Plecoptera, and Trichoptera, and (2) to separate Chironomidae from other  
119 Diptera, the most abundant taxon in our sampling. Other insects, mainly Diptera, were not  
120 considered because their contribution to total DM was low and did not justify the time-  
121 consuming work required for identification. For Chironomidae, sub-samples (1/2, 1/4 or 1/8)  
122 were obtained with a Motoda splitter (Motoda, 1959) when abundance was too high. Insects  
123 were then dried at 60 °C for at least 24 h. After cooling in a desiccator for another 24 h, each  
124 sample of each taxon was weighed to the nearest 0.2 mg (on a Mettler Toledo AG). Insect  
125 abundance ( $\text{ind m}^{-2} \text{d}^{-1}$ ) and DM ( $\text{mg DM m}^{-2} \text{d}^{-1}$ ) were recorded for each site separately and

126 then averaged (all sites combined) on a daily basis. Mean daily values obtained for each  
127 campaign were prolonged to adjacent days (both forward and backward) to fill temporal gaps  
128 between campaigns. Daily values were then summed to calculate cumulative seasonal  
129 emerging DM ( $\text{mg DM m}^{-2} \text{ season}^{-1}$ ) using solstice and equinox dates as reference. We  
130 calculated cumulative annual emerging DM as the sum of all seasonal values ( $\text{mg DM m}^{-2} \text{ y}^{-1}$ ).  
131 Biomass of emerging insects was then transformed to estimate secondary production at  
132 each site using the mean Emergence:Production (E:P) ratio of 0.183 calculated by Poepperl *et*  
133 *al.* (2000) for a similar agricultural stream. This ratio is relatively constant across taxa and  
134 stream ecosystems (Statzner and Resh 1993; Gratton and Zanden 2009).

#### 135 2.4 Water quality, land use and habitat assessment

136 Water quality data were provided for each study stream by the *Agence de l'Eau Loire-*  
137 *Bretagne* (2017). Information on water temperature ( $^{\circ}\text{C}$ ), turbidity (Formazin Nephelometric  
138 Units, FNU), conductivity ( $\mu\text{S cm}^{-1}$ ), nitrate ( $\text{mg NO}_3^- \text{ L}^{-1}$ ), ammonium ( $\text{mg NH}_4^+ \text{ L}^{-1}$ ),  
139 phosphorus (P) ( $\text{mg total P L}^{-1}$ ), suspended matter ( $\text{mg DM L}^{-1}$ ) and oxygen saturation (%)  
140 were collected to reflect water quality at the watershed scale. Monthly values observed from  
141 February 2016 to March 2017 were averaged to account for seasonal variations (Table S1).  
142 Similarly, mean annual discharge was estimated for each stream using QGIS (QGIS  
143 Development Team 2009), based on information available from the *Réseau Hydrographique*  
144 *Théorique* (Pella *et al.* 2012). Watershed-scale variables also included watershed area and  
145 land use (% of agricultural, artificial and natural area), which were estimated using QGIS and  
146 Corine Land Cover (CLC 2012). For each study site, local habitat features were measured or  
147 visually assessed at mid-channel every 5 m along a 20-m reach on both sides of each trap:  
148 surface water velocity ( $\text{cm s}^{-1}$ ), water depth (cm), depth of soft sediments on the bottom (cm),  
149 stream width (cm), bank height (cm), channel openness (%) and substrate size (Table S1).  
150 Substrate size was visually estimated following a modified Wentworth classification

151 (Cummins 1962). A substrate score was calculated by multiplying mean particulate size of  
152 each class by its observed proportion. Substrate heterogeneity was assessed by calculating the  
153 Shannon index of the proportions of each substrate size class. The embankment score (stream  
154 width divided by bank height) was used to estimate stream incision and erosion risk.

## 155 2.5 Statistical analyses

156 To study the temporal pattern of emergence, we fitted a linear mixed model (LMM) with a  
157 Gaussian distribution, considering the sampling campaign as a fixed effect. The campaign  
158 from 28 Nov to 5 Dec 2016 was ignored since data were available for only one site. To study  
159 effects of environmental variables on emerging insect DM, another LMM with a Gaussian  
160 distribution was applied. The small sample size precluded including both site- and watershed-  
161 scale variables in a single model. Consequently, two separate models were fitted to the dataset  
162 to assess effects of each set of variables.

163 The *corvif()* function of R software (R Core Team 2013) was used to verify correlations  
164 between variables before model computation (Zuur *et al.* 2009). Variables were removed  
165 when  $r$  coefficient of correlation exceeded 0.8 and the variance inflation factor exceeded 4.5.  
166 Accordingly, the following variables were selected and considered as fixed effects at the  
167 watershed scale: water temperature, turbidity, conductivity, oxygen saturation, ammonium  
168 nitrate concentrations and the percentage of watershed area covered by agriculture (cropland,  
169 pastures). At the site scale, the variables selected were water depth, stream width, substratum  
170 score, embankment score and channel openness. Prior to modeling, DM was log-transformed  
171 ( $\log x + 1$ ) to better meet assumptions of normality. Since quantitative explanatory variables  
172 were at different scales, we standardized them (mean=0,  $\sigma = 0.5$ ) using the *standardize()*  
173 function of the R package *arm* (Gelman A. and Hill J. 2007). Samplings were nested in  
174 *site/watershed* and *campaign*; these effects were thus included in the random part of temporal



175 and spatial models respectively. When a random effect was non-significant (*i.e.* when  
176 deviance explained was null), the model was re-run excluding it.

177 To identify variables that significantly influenced insect emergence, *model averaging* was  
178 preferred to a *best model* strategy. Indeed, *best model* strategy select only one model with one  
179 or several parameters and thus potentially exclude other model with different parameters  
180 having a similarly good fit (Whittingham *et al.* 2006). In an explanatory approach, *model*  
181 *averaging* is more appropriate as it produces estimates of parameters derived from multiple  
182 weighted models. The model-averaging method (Burnham and Anderson 2003) was applied  
183 at each spatial scale separately. Full models without interactions were used as starting models  
184 for the *dredge* function of the R package *MuMIn* (Barton K. 2016). We retained all models  
185 whose Akaike Information Criterion (AIC) corrected for small sample size (AICc; Grueber *et*  
186 *al.* 2011) differed by less than 4 from the best model. We also checked for correlation  
187 between fixed effects during the dredge process, but found no high correlation (all  $r < 0.7$ ).  
188 We performed model averaging on the selected subset using the *model.avg* function of the R  
189 package *MuMIn*. We considered explanatory variables significant when their associated 90%  
190 parameter confidence interval (CI) excluded zero (Arnold 2010). We assessed the significance  
191 of their effect on variations in DM of each taxon by considering their effect intensity through  
192 parameter estimates (*i.e.* effect size). Finally, the goodness-of-fit of all best averaged models  
193 ( $R^2$ ) was assessed using the procedure developed by Nakagawa and Schielzeth (2013) for  
194 mixed models using only significant explanatory variables.  $R^2_{\text{marginal}}$  approximates the  
195 variance explained by the fixed part of the model and  $R^2_{\text{conditional}}$  the variance explained by the  
196 complete model (fixed and random parts). All models were fitted using the R package *lme4*  
197 (Bates *et al.* 2015). The same datasets were used for temporal and environmental modeling.  
198 This allowed us to compare the AICc and  $R^2$  of models for each taxon.

## 199 3 Results

### 200 3.1 Variations in environmental characteristics

201 Stream discharge in 2016 peaked after heavy rainfall in March, two months before the first  
202 sampling campaign at all sites. Discharge rapidly decreased in April and May ( $< 0.5 \text{ m}^3 \text{ s}^{-1}$  for  
203 all streams), followed by a slight increase in June. No further rainfall increased stream  
204 discharge until February 2017, resulting in a low regime ( $< 0.1 \text{ m}^3 \text{ s}^{-1}$ ) for all streams from  
205 late summer to early winter. After some rainfall events in fall and winter, turbidity and  
206 suspended matter concentrations increased sharply to a mean of 16 FNU and  $15 \text{ mg DM L}^{-1}$ ,  
207 respectively. Mean water temperature ( $\pm 1$  standard deviation) ranged from  $17.4 \text{ }^\circ\text{C}$  ( $\pm 2.0 \text{ }^\circ\text{C}$ )  
208 in early summer to  $4.1 \text{ }^\circ\text{C}$  ( $\pm 1.5 \text{ }^\circ\text{C}$ ) in winter. Mean water conductivity was highest in fall  
209 ( $1,103 \text{ } \mu\text{S cm}^{-1} \pm 1,043$ ) and peaked at  $3,370 \text{ } \mu\text{S cm}^{-1}$  in Garun. Oxygen saturation ranged  
210 from 75% in summer to 100% in spring, with distinct annual variations among sites (Table  
211 S1). Mean nitrate concentration was high throughout the year ( $20.4 \text{ mg NO}_3^- \text{ L}^{-1} \pm 6.6$ ), with  
212 minimum values recorded at Illet and Veuvre and maximum values at Yaigne. Similarly,  
213 ammonium and total P concentrations varied greatly among streams. Agriculture dominates  
214 the landscape, covering a mean of 85% of the total watershed area, but variations in local  
215 habitat characteristics were observed among sites. For instance, the channel was narrow with  
216 dense riparian vegetation at Ormal but wider and almost completely open (no tree or shrub) at  
217 Roncelinain. At Tertre, the stream flowed very slowly ( $< 0.1 \text{ m s}^{-1}$ ), with an accumulation of  
218 fine sediment on the bottom, whereas at Jardière and Bray, water velocity, depth and  
219 substratum were more heterogeneous. Mean annual values of site- and watershed-scale  
220 variables are listed in Table S1.

## 221 3.2 Characteristics of aquatic insect emergence

222 We caught 64,027 insects in emergence traps during the study, equaling 13.1 g of total DM  
223 (all taxa combined). Chironomidae dominated total abundance (91%) and contributed 25% of  
224 total caught DM. Trichoptera contributed the most to total DM (56%) for only 6% of total  
225 abundance. Ephemeroptera represented 19% of total DM and 3% of total abundance.  
226 Plecoptera were rarely captured (only 23 individuals, 33.4 mg DM) and thus were not  
227 considered for further analysis. Total emerging DM caught in all sampling campaigns differed  
228 among sites and ranged from 435 (at Jardière) to 2,091 mg DM (at Vallée) (Table S2).

## 229 3.3 Temporal dynamics of insect emergence

230 Daily emergence varied by taxon among sampling campaigns (Fig. 1a). That of Trichoptera  
231 increased in spring and early summer (up to  $19.35 \text{ mg DM m}^{-2} \text{ d}^{-1}$ ), then decreased sharply  
232 and remained low until the end of the study ( $R^2_{\text{marginal}} = 34\%$ ). Daily emergence of  
233 Trichoptera varied greatly among sites in July, ranging from 2.62 to  $79.53 \text{ mg DM m}^{-2} \text{ d}^{-1}$  at  
234 Jardière and Fèvre, respectively (Table S3). Mean daily emergence of Chironomidae varied  
235 less among sampling campaigns ( $R^2_{\text{marginal}} = 11\%$ ), ranging from 2.89- $6.44 \text{ mg DM m}^{-2} \text{ d}^{-1}$ ,  
236 with two distinct emergence peaks in May and July. Ephemeroptera emergence decreased  
237 continuously ( $R^2_{\text{marginal}} = 34\%$ ) from spring (max.  $8.2 \text{ mg DM m}^{-2} \text{ d}^{-1}$ ) to winter. No  
238 emergence was observed for Ephemeroptera and Trichoptera in December, and it was very  
239 low for Chironomidae ( $\sim 0.17 \text{ mg DM m}^{-2} \text{ d}^{-1}$ ). Cumulative DM varied by taxon and season  
240 (Fig. 1b). Over the entire study, that of Trichoptera was highest ( $2,050 \text{ mg DM m}^{-2}$ ), followed  
241 by those of Chironomidae ( $1,120 \text{ mg DM m}^{-2}$ ) and Ephemeroptera ( $670 \text{ mg DM m}^{-2}$ ). For all  
242 taxa combined, cumulative DM reached  $3,840 \text{ mg DM m}^{-2} \text{ y}^{-1}$ , ranging from 1,445 to  $7,374$   
243  $\text{mg DM m}^{-2} \text{ y}^{-1}$  at Jardière and Fèvre, respectively. Using the Poepperl E:P ratio (2000), mean  
244 secondary production was estimated at  $21 \text{ g DM m}^{-2} \text{ y}^{-1}$ , ranging from 7.9 to  $40 \text{ g DM m}^{-2} \text{ y}^{-1}$   
245 at Jardière and Roncelinais, respectively.

### 246 3.4 Influence of environmental characteristics on insect emergence

247 The environmental variables in the spatial models explained a significant percentage of the  
248 variation in emerging insects DM, and noticeable differences were observed among taxa  
249 (Table 1). For Chironomidae, the spatial models performed better ( $R^2_{\text{marginal}} = 15.9\%$  and  
250  $21.4\%$  at the site and watershed scale, respectively) than the temporal model ( $R^2_{\text{marginal}} =$   
251  $10.9\%$ ). Among spatial models, the model with watershed-scale variables fit the data better  
252 but was less parsimonious than that with site-scale variables. Conversely, the temporal model  
253 explained emerging Trichoptera and Ephemeroptera DM best ( $R^2_{\text{marginal}} \sim 34\%$ ), the spatial  
254 models  $R^2_{\text{marginal}}$  ranging from 11.6 to 23.9%, with watershed-scale variables explaining  
255 Ephemeroptera better and site-scale variables explaining Trichoptera better.

256 Among the watershed-scale variables (Fig. 2a), the percentage of watershed area covered by  
257 agriculture positively and significantly influenced emerging Chironomidae DM. Emerging  
258 Chironomidae DM also increased significantly with increasing ammonium concentration and  
259 decreasing dissolved oxygen concentration. Conversely, emerging Ephemeroptera DM was  
260 significantly and negatively influenced by the percentage of watershed area covered by  
261 agriculture and, although not significantly, negatively influenced by ammonium concentration  
262 and positively influenced by dissolved oxygen concentration. Emerging Trichoptera and  
263 Ephemeroptera DM increased significantly with increasing water conductivity and  
264 temperature, but that of Trichoptera decreased with increasing turbidity. Among the site-scale  
265 variables (Fig. 2b), channel openness positively and significantly influenced emerging DM of  
266 all taxa. Emerging Trichoptera and Ephemeroptera DM increased significantly with  
267 decreasing stream incision (higher embankment scores), while that of Chironomidae increased  
268 significantly with decreasing stream width. Finally, emerging DM of Trichoptera increased  
269 significantly with increasing proportion of coarse substrate (*i.e.* higher substratum scores).

## 270 4 Discussion

271 We measured the biomass of aquatic insects emerging from third-order streams that drain  
272 agricultural landscapes in western France. Mean total emerging DM of the three dominant  
273 taxonomic groups living in temperate streams – Chironomidae, Ephemeroptera, and  
274 Trichoptera – equaled approximately  $4.0 \text{ g DM m}^{-2} \text{ y}^{-1}$ , ranging from  $1.4\text{-}7.4 \text{ g DM m}^{-2} \text{ y}^{-1}$   
275 depending on the stream. These results fall within the range of those reported in previous  
276 studies for emerging aquatic insects in streams (listed in Table 2), although exceptionally  
277 larger masses ( $> 20 \text{ g DM m}^{-2} \text{ y}^{-1}$ ) have been observed in desert streams in Arizona, United  
278 States (US) (Jackson and Fisher, 1986). In similar agricultural streams in Germany, Poepperl  
279 (2000) reported lower annual biomass of emerging aquatic insects, especially for Trichoptera  
280 and Ephemeroptera. The benthic biomass estimated by Gucker *et al.* (2011) was  
281 approximately 20% that observed in this study. The high biomass estimates in our streams are  
282 similar to those in other agriculture-dominated streams in several US states (Berg and  
283 Hellenthal 1991; Shieh *et al.* 2003; Johnson *et al.* 2013b). Compared to other forest-  
284 dominated streams, the biomass we estimated were four times greater for all taxa (combined)  
285 than in a third order stream in China (Yuen and Dudgeon 2016), but five times smaller for  
286 Chironomidae production than in a second order stream in Alabama (US) (Reynolds and  
287 Benke 2012).

288 The influence of water temperature on benthic invertebrate production and emergence (Huryn  
289 and Wallace 2000) was confirmed in our study by its positive correlation with Ephemeroptera  
290 and Trichoptera emergence, but not that of Chironomidae. Since Chironomidae species  
291 tolerate variations in water temperature better (Sponseller *et al.* 2001), the temperature range  
292 observed in the study may not have been wide enough to influence their biomass. This study  
293 confirmed that several agriculture-related variables can regulate the biomass of emerging  
294 stream insects (Allan *et al.* 1997). Indeed, proportion of agricultural land in catchment is often

295 associated with a decrease of emerging biomass of more sensitive taxa such as Ephemeroptera  
296 and Trichoptera (Piscart *et al.* 2009, 2011) and the enhancement of tolerant taxa such as  
297 Chironomidae (Lemly 1982). Sponseller *et al.* (2001) also reported that such pattern is  
298 correlated with a higher algal biomass in agricultural catchments of second to third order  
299 streams. At site scale in our study, the alteration of channel morphology by intensive  
300 agriculture practices, approximated by stream width and embankment score, was detrimental  
301 for Ephemeroptera and Trichoptera, while favoring Chironomidae biomass as reported in  
302 previous studies (Quinn and Hickey 1990, Kennedy and Turner 2011). Accumulation of silt is  
303 commonly reported in such altered streams (Wagenhoff *et al.* 2011). Filter-feeding taxa (*i.e.*  
304 Chironomidae and some Trichoptera) usually take advantage of fine sediment on the river  
305 bed, while shredders and scrapers (*i.e.* most Ephemeroptera and some Trichoptera) are  
306 hindered by these less favorable trophic conditions (Quinn and Hickey 1990, Compin and  
307 Céréghino 2007). Conversely, we observed that biomass of emerging Ephemeroptera and  
308 Trichoptera increased with increasing water conductivity. High conductivity is detrimental to  
309 benthic macroinvertebrate biomass and diversity (Johnson *et al.* 2013b), but high nutrient  
310 loads promote biofilm growth on stream substrate. Since biofilm is a major food source for  
311 Ephemeroptera and some Trichoptera, their biomass may have increased at sites with both  
312 high nutrient load and conductivity (Piscart *et al.* 2009). The positive correlation between  
313 channel openness and emergence further corroborates this hypothesis. Incident light enhances  
314 primary production in streams, favoring taxa that feed on algae and diatoms (Dance and  
315 Hynes 1980) in each taxonomic groups (e.g. *Baetis* or *Serratella* mayflies, Glossomatid  
316 caddisflies and many chironomids).

317 While Trichoptera emerged from early spring to summer, Ephemeroptera emerged only in  
318 spring. When sorting samples in the lab, we observed at least 15 morphospecies of  
319 Trichoptera (not taxonomically identified), suggesting that it had higher diversity than

320 Ephemeroptera. Since each species has its own timing of emergence, the longer emergence  
321 period for Trichoptera could have resulted simply from higher specific diversity.  
322 Chironomidae were caught almost continuously except in the coldest period in December,  
323 indicating that they emerged throughout the year, and were dominant taxa of emerging insects  
324 in fall, winter and early spring. These temporal patterns suggest that stream insects can  
325 subsidize surrounding agroecosystems for almost the entire year.

326 For accurate estimation of aquatic insect deposition on land, it is necessary to take into  
327 account the flux of winged aquatic insects returning to streams. Notably, females of several  
328 taxa lay their eggs at the surface or in water and died by drowning afterward (Baxter *et al.*  
329 2017). Nonetheless, several studies estimated that terrestrial mortality of female aquatic  
330 insects before oviposition is very high and exceeding 95% on average (reviewed by Huryn  
331 and Wallace 2000). This was reported for caddisfly, mortality of adult females reaching up to  
332 80% for Limnephilidae (Tricoptera, Enders and Wagner 1996) and 98.8% for Baetidae  
333 (Ephemeroptera, Werneke and Zick 1992), two families commonly found in agricultural  
334 streams (Friberg *et al.* 2009). These observations confirm other investigations on the low  
335 percentage of aquatic insects that could ultimately return to the stream (1-3%, Jackson and  
336 Fisher 1986, Gray 1989).

337 Based on the range of invertebrate production in our study, a stream section 100 m long  $\times$  6 m  
338 wide could produce 0.9-4.5 kg DM  $y^{-1}$  of emerging insects. Assuming that most winged  
339 aquatic insects disperse up to 50 m from the stream on each bank, aquatic insect deposition on  
340 land could reach 0.9-4.5 kg DM  $ha^{-1} y^{-1}$ . Using conversion factors of Evans-White *et al.*  
341 (2005), this could result in accumulation of 0.5-2.3, 0.1-0.5 and 0.005-0.03 kg  $ha^{-1} y^{-1}$  of  
342 carbon (C), nitrogen (N) and P on land, respectively, mainly in spring and summer. These  
343 subsidies are similar to reciprocal fluxes reported for other ecosystems (see Landry and  
344 Parrott 2016), notably the subsidies that carnivorous birds provide to freshwater or that deer

345 provide from fields to forests. In addition, terrestrial net secondary production can reach 1.9-  
346 4.0 kg C ha<sup>-1</sup> y<sup>-1</sup> in temperate grasslands (Gratton and Zanden 2009), *i.e.* approximately twice  
347 that expected along a stream 6 m wide. No estimate of secondary production exists for  
348 temperate cropland, but some studies highlighted that terrestrial insect communities are less  
349 diverse and abundant in agricultural areas (Benton *et al.* 2002; Batáry *et al.* 2012).

350 Taken together, these aspects suggest that aquatic subsidies provided by winged aquatic  
351 insects could play a significant role in agroecosystem functioning near streams. Chironomidae  
352 emerge throughout the year and can easily disperse over land (Muehlbauer *et al.* 2014),  
353 potentially providing a valuable source of nutrients (C, N, P) to receiving agroecosystems at  
354 critical times of the year. Recent studies highlighted the importance of aquatic insects in  
355 providing essential fatty acids to terrestrial consumers (Martin-Creuzburg *et al.* 2017; Popova  
356 *et al.* 2017). Many terrestrial predators (e.g. birds, Nakano and Murakami, 2001; spiders and  
357 carabids, Stenroth *et al.*, 2015) likely feed on aquatic prey and could later shift to terrestrial  
358 prey, including crop pests, as previously observed between fields and uncultivated areas  
359 (Rand *et al.* 2006).

## 360 5 Conclusion

361 Multiple influences on aquatic insect emergence were identified at the local and watershed  
362 scales, and some were directly related to agricultural practices. Our results suggest that  
363 emergence and dispersal of aquatic insects could substantially influence terrestrial ecosystem  
364 functioning in landscapes with a dense waterway network. More studies are required on the  
365 dispersal of aquatic winged insects and their deposition in terrestrial ecosystems to better  
366 understand and quantify the ecological processes involved. This will help to assess effects of  
367 the aquatic subsidies that streams provide on ecosystem services supplied to agriculture, such  
368 as fertilization, biological control and pollination.



369 *Acknowledgement*

370 This work received financial support from the Metaprogramm EcoServ of the French National  
371 Institute for Agronomical Research (INRA) to JM Roussel, and from the French Ministry for  
372 Reseach (PhD grant 2015-2018) to J Raitif. We thank two anonymous reviewers for their  
373 constructive comments on the first version of the manuscript.

374 References

- 375 Agence de l'eau Loire-Bretagne (2017) Base de données de bassin : OSUR. [http://osur.eau-](http://osur.eau-loire-bretagne.fr/exportosur/Accueil)  
376 [loire-bretagne.fr/exportosur/Accueil](http://osur.eau-loire-bretagne.fr/exportosur/Accueil). Accessed 1 Oct 2017
- 377 Allan D, Erickson D, Fay J (1997) The influence of catchment land use on stream integrity  
378 across multiple spatial scales. *Freshw Biol* 37:149–161. doi: 10.1046/j.1365-  
379 2427.1997.d01-546.x
- 380 Arnold TW (2010) Uninformative Parameters and Model Selection Using Akaike's  
381 Information Criterion. *J Wildl Manag* 74:1175–1178. doi: 10.1111/j.1937-  
382 2817.2010.tb01236.x
- 383 Bartels P, Cucherousset J, Steger K, et al (2012) Reciprocal subsidies between freshwater and  
384 terrestrial ecosystems structure consumer resource dynamics. *Ecology* 93:1173–1182
- 385 Barton K. (2016) MuMIn: multi-model inference. R package version 1.15.6. [http://cran.r-](http://cran.r-project.org/web/packages/MuMIn/index.html)  
386 [project.org/web/packages/MuMIn/index.html](http://cran.r-project.org/web/packages/MuMIn/index.html)
- 387 Batáry P, Holzschuh A, Orci KM, et al (2012) Responses of plant, insect and spider  
388 biodiversity to local and landscape scale management intensity in cereal crops and  
389 grasslands. *Agric Ecosyst Environ* 146:130–136. doi: 10.1016/j.agee.2011.10.018
- 390 Bates D, Maechler M., Bolker B., Walker S. (2015) Fitting Linear Mixed-Effects Models  
391 Using lme4. *Journal of Statistical Software* 67:1–48. doi: 10.18637/jss.v067.i01
- 392 Baxter CV, Kennedy TA, Miller SW, et al (2017) Macroinvertebrate Drift, Adult Insect  
393 Emergence and Oviposition. In: *Methods in Stream Ecology, Volume 1*. Elsevier, pp  
394 435–456
- 395 Beketov MA, Kefford BJ, Schafer RB, Liess M (2013) Pesticides reduce regional biodiversity  
396 of stream invertebrates. *Proc Natl Acad Sci* 110:11039–11043. doi:  
397 10.1073/pnas.1305618110
- 398 Benton TG, Bryant DM, Cole L, Crick HQP (2002) Linking agricultural practice to insect and  
399 bird populations: a historical study over three decades: Farming, insect and bird  
400 populations. *J Appl Ecol* 39:673–687. doi: 10.1046/j.1365-2664.2002.00745.x
- 401 Berg MB, Hellenthal RA (1991) Secondary production of Chironomidae (Diptera) in a north  
402 temperate stream. *Freshw Biol* 25:497–505. doi: 10.1111/j.1365-2427.1991.tb01392.x
- 403 Burnham KP, Anderson DR (2003) *Model Selection and Multimodel Inference: A Practical*  
404 *Information-Theoretic Approach*. Springer Science & Business Media
- 405 Carlson PE, McKie BG, Sandin L, Johnson RK (2016) Strong land-use effects on the  
406 dispersal patterns of adult stream insects: implications for transfers of aquatic  
407 subsidies to terrestrial consumers. *Freshwater Biology* 61:848–861. doi:  
408 10.1111/fwb.12745
- 409 CLC (2012) Copernicus land monitoring services. [http://land.copernicus.eu/pan-](http://land.copernicus.eu/pan-european/corine-land-cover/clc-2012)  
410 [european/corine-land-cover/clc-2012](http://land.copernicus.eu/pan-european/corine-land-cover/clc-2012). Accessed 13 Jul 2016

- 411 Compin A, Céréghino R (2007) Spatial patterns of macroinvertebrate functional feeding  
412 groups in streams in relation to physical variables and land-cover in Southwestern  
413 France. *Landscape Ecology* 22:1215–1225. doi: 10.1007/s0980-007-9101-y
- 414 Corbet PS (1964) Temporal Patterns of Emergence in Aquatic Insects. *Can Entomol* 96:264–  
415 279. doi: 10.4039/Ent96264-1
- 416 Cross WF, Wallace JB, Rosemond AD, Eggert SL (2006) Whole-system nutrient enrichment  
417 increases secondary production in a detritus-based ecosystem. *Ecology* 87:1556–1565.  
418 doi: 10.1890/0012-9658(2006)87[1556:WNEISP]2.0.CO;2
- 419 Cummins KW (1962) An Evaluation of Some Techniques for the Collection and Analysis of  
420 Benthic Samples with Special Emphasis on Lotic Waters. *Am Midl Nat* 67:477–504.  
421 doi: 10.2307/2422722
- 422 Dance KW, Hynes HBN (1980) Some effects of agricultural land use on stream insect  
423 communities. *Environmental Pollution Series A, Ecological and Biological* 22:19–28.  
424 doi: 10.1016/0143-1471(80)90078-1
- 425 Davis JM, Rosemond AD, Eggert SL, et al (2010) Nutrient enrichment differentially affects  
426 body sizes of primary consumers and predators in a detritus-based stream. *Limnol*  
427 *Oceanogr* 55:2305–2316. doi: 10.4319/lo.2010.55.6.2305
- 428 Deegan BM, Ganf GG (2008) The loss of aquatic and riparian plant communities:  
429 Implications for their consumers in a riverine food web. *Austral Ecol* 33:672–683. doi:  
430 10.1111/j.1442-9993.2008.01834.x
- 431 Dreyer J, Townsend PA, Hook JC III, et al (2015) Quantifying aquatic insect deposition from  
432 lake to land. *Ecology* 96:499–509. doi: 10.1890/14-0704.1
- 433 Dudgeon D, Arthington AH, Gessner MO, et al (2006) Freshwater biodiversity: importance,  
434 threats, status and conservation challenges. *Biol Rev* 81:163-182. doi:  
435 10.1017/s464793105006950
- 436 Enders G, Wagner R (1996) Mortality of *Apatania fimbriata* (Insecta: Trichoptera) during  
437 embryonic, larval and adult life stages. *Freshwater Biology* 36:93–104. doi:  
438 10.1046/j.1365-2427.1996.00081.x
- 439 Evans-White MA, Stelzer RS, Lamberti GA (2005) Taxonomic and regional patterns in  
440 benthic macroinvertebrate elemental composition in streams. *Freshw Biol* 50:1786–  
441 1799. doi: 10.1111/j.1365-2427.2005.01455.x
- 442 Friberg N, Skriver J, Larsen SE, et al (2009) Stream macroinvertebrate occurrence along  
443 gradients in organic pollution and eutrophication: Macroinvertebrate response to  
444 organic pollution and eutrophication. *Freshwater Biology* 55:1405–1419. doi:  
445 10.1111/j.1365-2427.2008.02164.x
- 446 Gelman A., Hill J. (2007) *Data Analysis Using Regression and Multilevel/Hierarchical*  
447 *Models*. Cambridge University Press
- 448 Gratton C, Zanden MJV (2009) Flux of aquatic insect productivity to land: comparison of  
449 lentic and lotic ecosystems. *Ecology* 90:2689–2699. doi: 10.1890/08-1546.1

- 450 Gray LJ (1989) Emergence Production and Export of Aquatic Insects from a Tallgrass Prairie  
451 Stream. *The Southwestern Naturalist* 34:313–318. doi: 10.2307/36721588
- 452 Gray LJ (1993) Response of Insectivorous Birds to Emerging Aquatic Insects in Riparian  
453 Habitats of a Tallgrass Prairie Stream. *Am Midl Nat* 129:288-300. doi:  
454 10.2307/2426510
- 455 Greenwood MJ, Booker DJ (2016) Influence of hydrological regime and land cover on traits  
456 and potential export capacity of adult aquatic insects from river channels. *Oecologia*  
457 180:551–566. doi: 10.1007/s00442-015-3462-8
- 458 Grueber CE, Nakagawa S, Laws RJ, Jamieson IG (2011) Multimodel inference in ecology  
459 and evolution: challenges and solutions: Multimodel inference. *J Evol Biol* 24:699–  
460 711. doi: 10.1111/j.1420-9101.2010.02210.x
- 461 Gücker B, Brauns M, Solimini AG, et al (2011) Urban stressors alter the trophic basis of  
462 secondary production in an agricultural stream. *Can J Fish Aquat Sci* 68:74–88. doi:  
463 10.1139/F10-126
- 464 Hanna DEL, Tomscha SA, Ouellet Dallaire C, Bennett EM (2017) A review of riverine  
465 ecosystem service quantification: research gaps and recommendations. *J Appl Ecol*.  
466 doi: 10.1111/1365-2664.13045
- 467 Havik G, Catenazzi A, Holmgren M (2014) Seabird Nutrient Subsidies Benefit Non-Nitrogen  
468 Fixing Trees and Alter Species Composition in South American Coastal Dry Forests.  
469 *PLoS ONE* 9:e86381. doi: 10.1371/journal.pone.0086381
- 470 Hering D, Plachter H (1997) Riparian ground beetles (Coeloptera, Carabidae) preying on  
471 aquatic invertebrates: a feeding strategy in alpine floodplains. *Oecologia* 111:261–270.  
472 doi: 10.1007/s004420050234
- 473 Huryn AD, Wallace JB (2000) Life History and Production of Stream Insects. *Annu Rev*  
474 *Entomol* 45:83–110. doi: 10.1146/annurev.ento.45.1.83
- 475 Jackson JK, Fisher SG (1986) Secondary Production, Emergence, and Export of Aquatic  
476 Insects of a Sonoran Desert Stream. *Ecology* 67:629-638. doi: 10.2307/1937686
- 477 Johnson BR, Fritz KM, Price R (2013a) Estimating benthic secondary production from  
478 aquatic insect emergence in streams affected by mountaintop removal coal mining,  
479 West Virginia, USA. *Fundam Appl Limnol Arch Für Hydrobiol* 182/3:191–204. doi:  
480 10.1127/1863-9135/2013/0403
- 481 Johnson RC, Jin H-S, Carreiro MM, Jack JD (2013b) Macroinvertebrate community structure,  
482 secondary production and trophic-level dynamics in urban streams affected by  
483 non-point-source pollution. *Freshw Biol* 58:843–857. doi: 10.1111/fwb.12090
- 484 Kennedy TL, Turner TF (2011) River channelization reduces nutrient flow and  
485 macroinvertebrate diversity at the aquatic terrestrial transition zone. *Ecosphere*  
486 2:art35. doi: 10.1890/ES11-00047.1
- 487 Landry J-S, Parrott L (2016) Could the lateral transfer of nutrients by outbreaking insects lead  
488 to consequential landscape-scale effects? *Ecosphere* 7:e01265. doi: 10.1002/ecs2.1265

- 489 Lemly AD (1982) Modification of benthic insect communities in polluted streams: combined  
 490 effects of sedimentation and nutrient enrichment. *Hydrobiologia* 87:229–245. doi:  
 491 10.1007/BF00007232
- 492 Liess M, Von Der Ohe PC (2005) Analysing effects of pesticides on invertebrate communities  
 493 in streams. *Environ Toxicol Chem* 24:954–965. doi: 10.1897/03-652.1
- 494 Magbanua FS, Townsend CR, Hageman KJ, et al (2016) Individual and combined effects of  
 495 fine sediment and glyphosate herbicide on invertebrate drift and insect emergence: a  
 496 stream mesocosm experiment. *Freshw Sci* 35:139–151. doi: 10.1086/684363
- 497 Martin-Creuzburg D, Kowarik C, Straile D (2017) Cross-ecosystem fluxes: Export of  
 498 polyunsaturated fatty acids from aquatic to terrestrial ecosystems via emerging insects.  
 499 *Sci Total Environ* 577:174–182. doi: 10.1016/j.scitotenv.2016.10.156
- 500 Météo France (2017) Données Climatiques de la station de Rennes. source :  
 501 <http://www.meteofrance.com/climat/france/rennes/35281001/normales>. Accessed 13  
 502 Nov 2017
- 503 Motoda, S. 1959. Devices of simple plankton apparatus. *Memoirs of the Faculty of Fisheries,*  
 504 *Hokkaido University*, 1:73–94.
- 505 Muehlbauer JD, Collins SF, Doyle MW, Tockner K (2014) How wide is a stream? Spatial  
 506 extent of the potential “stream signature” in terrestrial food webs using meta-analysis.  
 507 *Ecology* 95:44–55. doi: 10.1890/12-1628.1
- 508 Nagasaka A, Nakamura F (1999) The influences of land-use changes on hydrology and  
 509 riparian environment in a northern Japanese landscape. *Landsc Ecol* 14:543–556. doi:  
 510 10.1023/A:1008164123102
- 511 Naiman RJ and Décamps H (1997) THE ECOLOGY OF INTERFACES: Riparian Zones.  
 512 *Annu Rev Ecol Syst* 28:621–658. doi: 10.1146/annurev.ecolsys.28.1.621
- 513 Nakagawa S, Schielzeth H (2013) A general and simple method for obtaining  $R^2$  from  
 514 generalized linear mixed-effects models. *Methods in Ecology and Evolution* 4:133–  
 515 142. doi: 10.1111/j.2041-210x.2012.00261.x
- 516 Nakano S, Murakami M (2001) Reciprocal subsidies: Dynamic interdependence between  
 517 terrestrial and aquatic food webs. *Proc Natl Acad Sci U S A* 98:166–170
- 518 Paetzold A, Schubert CJ, Tockner K (2005) Aquatic Terrestrial Linkages Along a Braided-  
 519 River: Riparian Arthropods Feeding on Aquatic Insects. *Ecosystems* 8:748–759. doi:  
 520 10.1007/s0021-005-0004-y
- 521 Pella, H., Lejot, J., Lamouroux, N. 2012. Le réseau hydrographique théorique (RHT) français  
 522 et ses attributs environnementaux, *Géomorphologie : relief, processus, environnement*
- 523 Piscart C, Genoel R, Doledec S, et al (2009) Effects of intense agricultural practices on  
 524 heterotrophic processes in streams. *Environ Pollut* 157:1011–1018. doi:  
 525 10.1016/j.envpol.2008.10.010

- 526 Piscart C, Navel S, Maazouzi C, et al (2011) Leaf litter recycling in benthic and hyporheic  
527 layers in agricultural streams with different types of land use. *Sci Total Environ*  
528 409:4373–4380. doi: 10.1016/j.scitotenv.2011.06.060
- 529 Poepperl R (2000) Benthic secondary production and biomass of insects emerging from a  
530 northern German temperate stream. *Freshw Biol* 44:199–211. doi: 10.1046/j.1365-  
531 2427.2000.00558.x
- 532 Polis GA, Anderson WB, Holt RD (1997) Toward an integration of landscape and food web  
533 ecology: the dynamics of spatially subsidized food webs. *Annual review of ecology*  
534 and systematics 289–316
- 535 Popova ON, Haritonov AY, Sushchik NN, et al (2017) Export of aquatic productivity,  
536 including highly unsaturated fatty acids, to terrestrial ecosystems via Odonata. *Sci*  
537 *Total Environ* 581-582:40–48. doi: 10.1016/j.scitotenv.2017.01.017
- 538 QGIS Development Team, 2009. QGIS Geographic Information System. Open Source  
539 Geospatial Foundation. URL <http://qgis.osgeo.org>
- 540 Quinn JM, Hickey CW (1990) Characterisation and classification of benthic invertebrate  
541 communities in 88 New Zealand rivers in relation to environmental factors. *New*  
542 *Zealand Journal of Marine and Freshwater Research* 24:387–409. doi:  
543 10.1080/00288330.1990.9516432
- 544 Rabení CF, Doisy KE, Zweig LD (2005) Stream invertebrate community functional responses  
545 to deposited sediment. *Aquat Sci* 67:395–402. doi: 10.1007/s00027-005-0793-2
- 546 Raitif J, Plantegenest M, Piscart C, Roussel J-M (in revision) From stream to land: ecosystem  
547 services provided by stream insects to agriculture. *Agric Ecosyst Environ*
- 548 Rand TA, Tylianakis JM, Tschardt T (2006) Spillover edge effects: the dispersal of  
549 agriculturally subsidized insect natural enemies into adjacent natural habitats. *Ecol*  
550 *Lett* 9:603–614. doi: 10.1111/j.1461-0248.2006.00911.x
- 551 R Core Team (2013) R: A language and environment for statistical computing. R Foundation  
552 for Statistical Computing, Vienna, Austria
- 553 Reynolds SK, Benke AC (2012) Chironomid production along a hyporheic gradient in  
554 contrasting stream types. *Freshw Sci* 31:167–181. doi: 10.1899/11-017.1
- 555 Richardson JS, Zhang Y, Marczak LB (2010) Resource subsidies across the land-freshwater  
556 interface and responses in recipient communities. *River Res Appl* 26:55–66. doi:  
557 10.1002/rra.1283
- 558 Sallenave RM, Day KE (1991) Secondary production of benthic stream invertebrates in  
559 agricultural watersheds with different land management practices. *Chemosphere*  
560 23:57–76. doi: 10.1016/0045-6535(91)90116-U
- 561 Shieh S-H, Ward JV, Kondratieff BC (2003) Longitudinal changes in macroinvertebrate  
562 production in a stream affected by urban and agricultural activities. *Arch Für*  
563 *Hydrobiol* 157:483–503. doi: 10.1127/0003-9136/2003/0157-0483



- 564 Sponseller RA, Benfield EF, Valett HM (2001) Relationships between land use, spatial scale  
565 and stream macroinvertebrate communities. *Freshw Biol* 46:1409–1424. doi:  
566 10.1046/j.1365-2427.2001.00758.x
- 567 Statzner B, Resh VH (1993) Multiple-site and-year analyses of stream insect emergence: a  
568 test of ecological theory. *Oecologia* 96:65–79. doi: 10.1007/BF00318032
- 569 Stenroth K, Polvi LE, Fältström E, Jonsson M (2015) Land-use effects on terrestrial  
570 consumers through changed size structure of aquatic insects. *Freshw Biol* 60:136–149.  
571 doi: 10.1111/fwb.12476
- 572 Tachet H, Richoux P, Bournaud M, Usseglio-Polatera P (2010) Invertébrés d’eau douce  
573 systématique, biologie, écologie. CNRS éd., Paris
- 574 Wagenhoff A, Townsend CR, Phillips N, Matthaei CD (2011) Subsidy-stress and multiple-  
575 stressor effects along gradients of deposited fine sediment and dissolved nutrients in a  
576 regional set of streams and rivers: Sediment and nutrients in streams. *Freshw Biol*  
577 56:1916–1936. doi: 10.1111/j.1365-2427.2011.02619.x
- 578 Werneke U, Zwick P (1992) Mortality of the terrestrial adult and aquatic nymphal life stages  
579 of *Baetis vernalis* and *Baetis rhodani* in the Breitenbach, Germany (Insecta:  
580 Ephemeroptera). *Freshwater Biology* 28:249–255. doi: 10.1111/j.1365-  
581 2427.1992.tb00581.x
- 582 Whittingham MJ, Stephens PA, Bradbury RB, Freckleton RP (2006) Why do we still use  
583 stepwise modelling in ecology and behaviour? *Journal of Animal Ecology* 75:1182–  
584 1189. doi: 10.1111/j.1365-2656.2006.01141.x
- 585 Yuen EYL, Dudgeon D (2016) The magnitude and seasonality of aquatic insect subsidies to  
586 tropical stream riparia in Hong Kong. *Aquat Sci* 78:655–667. doi: 10.1007/s00027-  
587 015-0455-y
- 588 Zuur A, Ieno EN, Walker N, et al (2009) *Mixed Effects Models and Extensions in Ecology*  
589 with R. Springer-Verlag, New York

Table 1. Comparison of temporal and spatial fitted mix-models for each taxon. Summarizing statistics are: Akaike Information Criterion score corrected for small sample size ( $AIC_c$ ), the variance explained by fixed parameters (marginal  $R^2$ ) and both fixed and random parameters (conditional  $R^2$ ) of the model. Null models are intercept-only models, which have no fixed parameter and the same random structure as fitted models. They are used as bases of comparison against fitted models for  $AIC_c$  and  $R^2$ . For a given taxon and scale (*i.e.* temporal or spatial), lower  $AIC_c$  indicates a better fit of the model to the data, higher marginal/conditional  $R^2$  indicate greater goodness-of-fit (the amount of variance explained).

Taxon	Statistic	Temporal model		Spatial models		
		Null	Sampling campaign	Null	Watershed	Site
Chironomidae	$AIC_c$	305.1	298.6	301.8	294.9	293.5
	Marginal/conditional $R^2$	0.0/22.0	10.9/34.9	0.0/32.7	21.4/32.5	15.9/41.9
Trichoptera	$AIC_c$	323.6	285.4	294.3	292.2	272.0
	Marginal/conditional $R^2$	0.0/13.3	34.1/52.4	0.0/52.5	11.6/50.5	23.9/57.2
Ephemeroptera	$AIC_c$	225.3	196.1	204.2	195.7	201.7
	Marginal/conditional $R^2$	0.0/11.4	33.9/52.1	0.0/51.8	18.8/50.9	11.9/43.5



Table 2. Mean annual dry mass (with range of values in brackets when available) of emerging aquatic insect or secondary production. Values in italics correspond to estimates based on the Poepperl (2000) Emergence:Production ratio. EPT = Ephemeroptera, Plecoptera and Trichoptera.

References	Location	Stream order or width	Landscape	Annual emergence $\text{g m}^{-2}$	Annual secondary production $\text{g m}^{-2}$	Taxa
Jackson & Fisher 1986	Arizona, USA	3-4 m	Desert	13.8	58.3	Chironomidae
				4.4	55.5	Ephemeroptera
				2.9	3.6	Trichoptera
Berg & Hellenthal 1991	Indiana, USA	third order	Agriculture/Forest	5.4	29.7	Chironomidae
Sallenave et Day, 1991	Ontario, Canada	2-4 m	Agriculture/Forest	2.9 [1.1-4.7]	15.6 [5.8-25.4]	Trichoptera
Poepperl 2000	Schleswig-Holstein, Germany	6 m	Agriculture	1.1 [0.6-1.3]	5.3 [3-6.1]	Diptera
				0.1 [0.01-0.1]	0.6 [<0.1-0.9]	Ephemeroptera
				0.5 [0.3-0.6]	1.9 [1-2.1]	Trichoptera
Shieh <i>et al.</i> 2003	Colorado, USA	fifth order	Agriculture	4.2 [3.3-5]	23.2 [17.9-27.2]	Chironomidae
				0.4 [0-0.7]	2.4 [0.02-3.7]	Ephemeroptera
				1.7 [0.4-2.2]	9.5 [2.3-12]	Trichoptera
Gücker <i>et al.</i> 2011	Berlin, Germany	3-4 m	Agriculture	-	3.9 [0.4-70.9]	EPT + Chironomidae + other taxa
Reynolds & Benke 2012	Alabama, USA	second order	Forest	5.3 [1.4-12.5]	29.2 [7.8 - 68.3]	Chironomidae
Johnson <i>et al.</i> 2013	Kentucky, USA	6 m	Agriculture/Forest	-	21.8	EPT + Chironomidae + other taxa
		6 m	Agriculture/Forest	1.4	7.5	EPT
Yuen & Dudgeon 2016	Hong Kong, China	third or fourth order	Forest	0.9 [0.7-1.2]	4.9 [3.8-6.6]	EPT + Chironomidae + other taxa
This study	Brittany, France	third order	Agriculture	3.8 [1.4-7.4]	21 [7.9-40.2]	EPT + Chironomidae
				1.1 [0.2-2.2]	6.1 [1.3-11.8]	Chironomidae
				0.7 [0.1-1.6]	3.7 [0.4-7.6]	Ephemeroptera
				2.1 [0.7-4.9]	11.2 [2.9-24.8]	Trichoptera

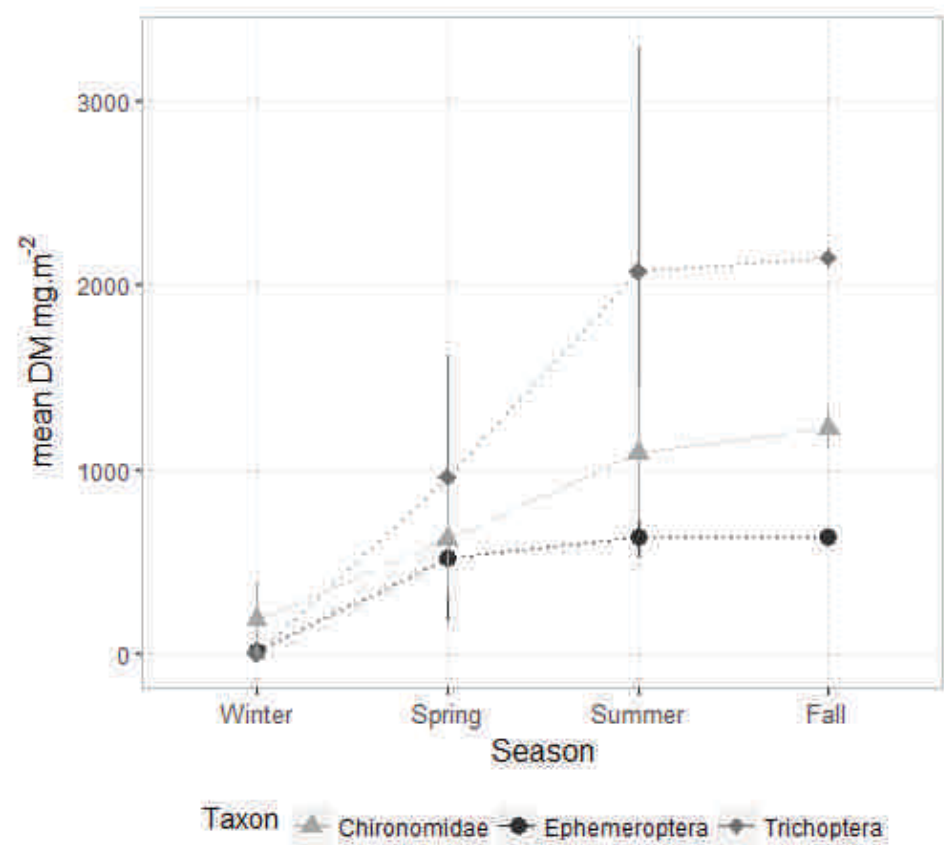
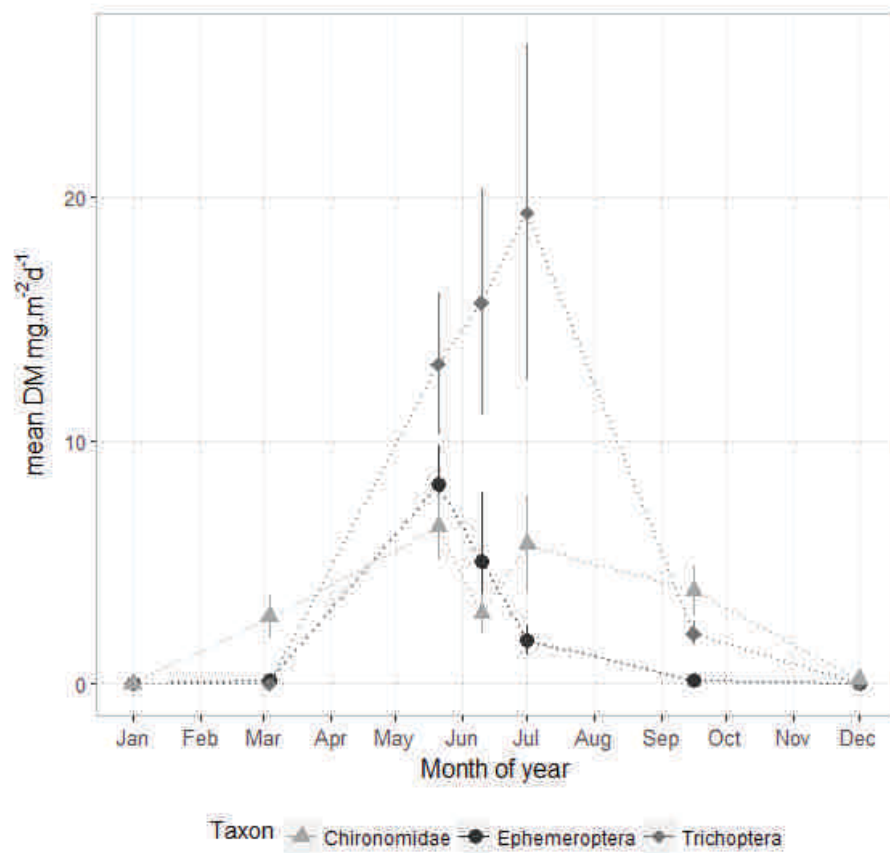


Figure 1. Emergence of aquatic insects dry mass (DM) all sites combined. Left panel shows mean ( $\pm 1$  standard deviation SD) daily values for each sampling date and taxonomic group. Right panel shows the mean ( $\pm 1$  SD) cumulative emergence by season (see text for details).

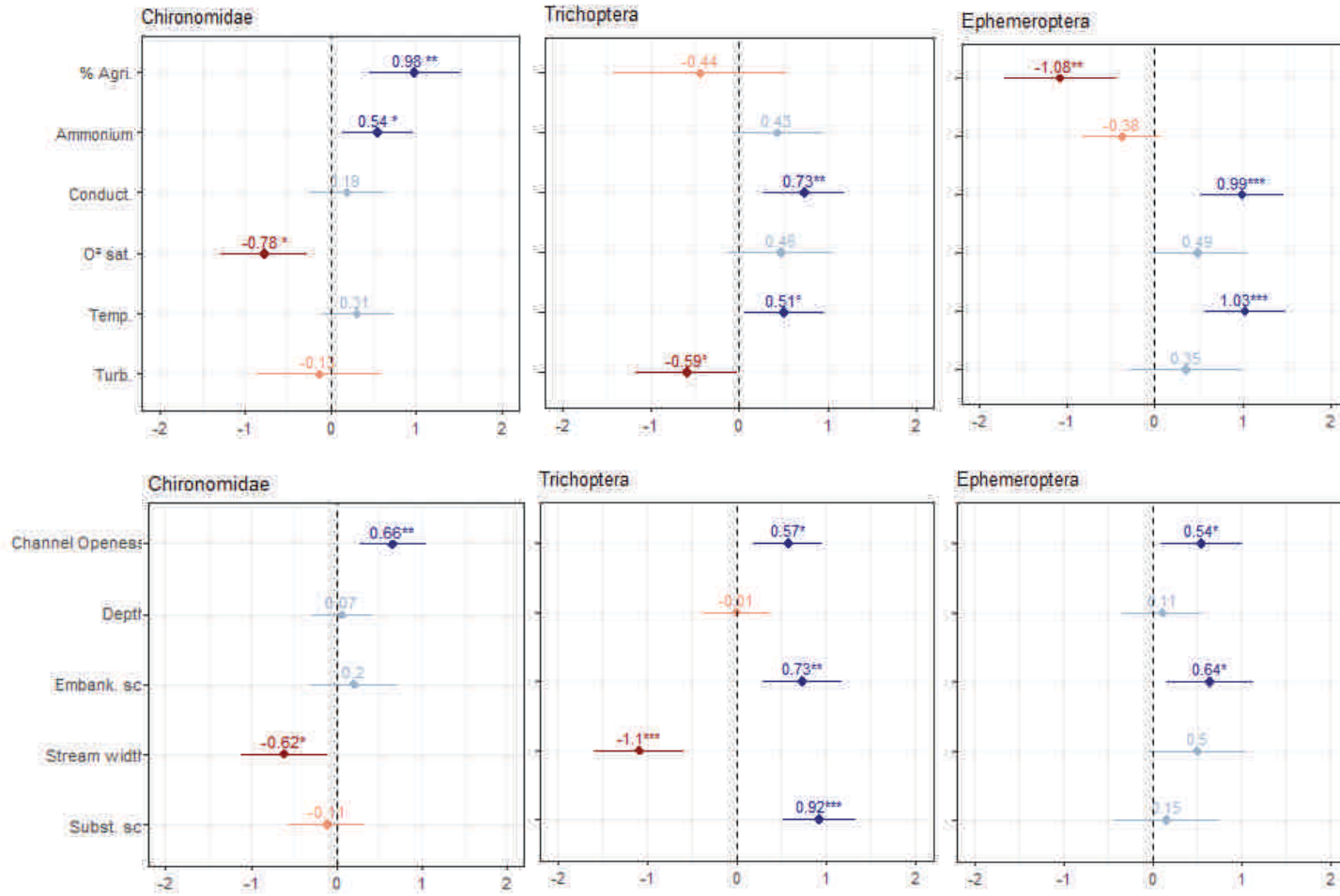


Figure 2. Standardized effect of environmental variables (mean effect size  $\pm$  90% confidence interval) at watershed and site scales on emerging biomass of Chironomidae (left), Trichoptera (middle) and Ephemeroptera (right). Upper panel: Watershed-scale variables; % Agri = Percentage of watershed area covered by agriculture (cropland and pasture), Ammonium = Ammonium (NH<sub>4</sub><sup>+</sup>) concentration in water, Conductivity =

Water conductivity, O<sup>2</sup> Sat. = Oxygen saturation, Temp. = Water temperature, Turb. = Water turbidity. Lower panel: Site-scale variables;

Channel Openness = Proportion of non-canopy cover over the stream channel, Depth = Water depth, Embank. Sc. = Stream width divided by bank height, Stream width = channel width, Subst. sc. = Substratum score, based on modified Wenworth classification (Cummins, 1962).

Colors indicate degree of positive or negative effect. ° =  $p < 0.10$ , \* =  $p < 0.05$ , \*\* =  $p < 0.01$ , \*\*\* =  $p < 0.001$

Table 1. Comparison of temporal and spatial fitted mix-models for each taxon. Summarizing statistics are: Akaike Information Criterion score corrected for small sample size ( $AIC_c$ ), the variance explained by fixed parameters (marginal  $R^2$ ) and both fixed and random parameters (conditional  $R^2$ ) of the model. Null models are intercept-only models, which have no fixed parameter and the same random structure as fitted models. They are used as bases of comparison against fitted models for  $AIC_c$  and  $R^2$ . For a given taxon and scale (*i.e.* temporal or spatial), lower  $AIC_c$  indicates a better fit of the model to the data, higher marginal/conditional  $R^2$  indicate greater goodness-of-fit (the amount of variance explained).

Taxon	Statistic	Temporal model		Spatial models		
		Null	Sampling campaign	Null	Watershed	Site
Chironomidae	$AIC_c$	305.1	298.6	301.8	294.9	293.5
	Marginal/conditional $R^2$	0.0/22.0	10.9/34.9	0.0/32.7	21.4/32.5	15.9/41.9
Trichoptera	$AIC_c$	323.6	285.4	294.3	292.2	272.0
	Marginal/conditional $R^2$	0.0/13.3	34.1/52.4	0.0/52.5	11.6/50.5	23.9/57.2
Ephemeroptera	$AIC_c$	225.3	196.1	204.2	195.7	201.7
	Marginal/conditional $R^2$	0.0/11.4	33.9/52.1	0.0/51.8	18.8/50.9	11.9/43.5

Table 2. Mean annual dry mass (with range of values in brackets when available) of emerging aquatic insect or secondary production. Values in italics correspond to estimates based on the Poepperl (2000) Emergence:Production ratio. EPT = Ephemeroptera, Plecoptera and Trichoptera.

References	Location	Stream order or width	Landscape	Annual emergence $\text{g m}^{-2}$	Annual secondary production $\text{g m}^{-2}$	Taxa
Jackson & Fisher 1986	Arizona, USA	3-4 m	Desert	13.8	58.3	Chironomidae
				4.4	55.5	Ephemeroptera
				2.9	3.6	Trichoptera
Berg & Hellenthal 1991	Indiana, USA	third order	Agriculture/Forest	5.4	29.7	Chironomidae
Sallenave et Day, 1991	Ontario, Canada	2-4 m	Agriculture/Forest	2.9 [1.1-4.7]	15.6 [5.8-25.4]	Trichoptera
Poepperl 2000	Schleswig-Holstein, Germany	6 m	Agriculture	1.1 [0.6-1.3]	5.3 [3-6.1]	Diptera
				0.1 [0.01-0.1]	0.6 [<0.1-0.9]	Ephemeroptera
				0.5 [0.3-0.6]	1.9 [1-2.1]	Trichoptera
Shieh <i>et al.</i> 2003	Colorado, USA	fifth order	Agriculture	4.2 [3.3-5]	23.2 [17.9-27.2]	Chironomidae
				0.4 [0-0.7]	2.4 [0.02-3.7]	Ephemeroptera
				1.7 [0.4-2.2]	9.5 [2.3-12]	Trichoptera
Gücker <i>et al.</i> 2011	Berlin, Germany	3-4 m	Agriculture	-	3.9 [0.4-70.9]	EPT + Chironomidae + other taxa
Reynolds & Benke 2012	Alabama, USA	second order	Forest	5.3 [1.4-12.5]	29.2 [7.8 - 68.3]	Chironomidae
Johnson <i>et al.</i> 2013	Kentucky, USA	6 m	Agriculture/Forest	-	21.8	EPT + Chironomidae + other taxa
		6 m	Agriculture/Forest	1.4	7.5	EPT
Yuen & Dudgeon 2016	Hong Kong, China	third or fourth order	Forest	0.9 [0.7-1.2]	4.9 [3.8-6.6]	EPT + Chironomidae + other taxa
This study	Brittany, France	third order	Agriculture	3.8 [1.4-7.4]	21 [7.9-40.2]	EPT + Chironomidae
				1.1 [0.2-2.2]	6.1 [1.3-11.8]	Chironomidae
				0.7 [0.1-1.6]	3.7 [0.4-7.6]	Ephemeroptera
				2.1 [0.7-4.9]	11.2 [2.9-24.8]	Trichoptera

**Figure**

[Click here to download Figure: Raitif-et-al-v2.0\\_Figures.docx](#)

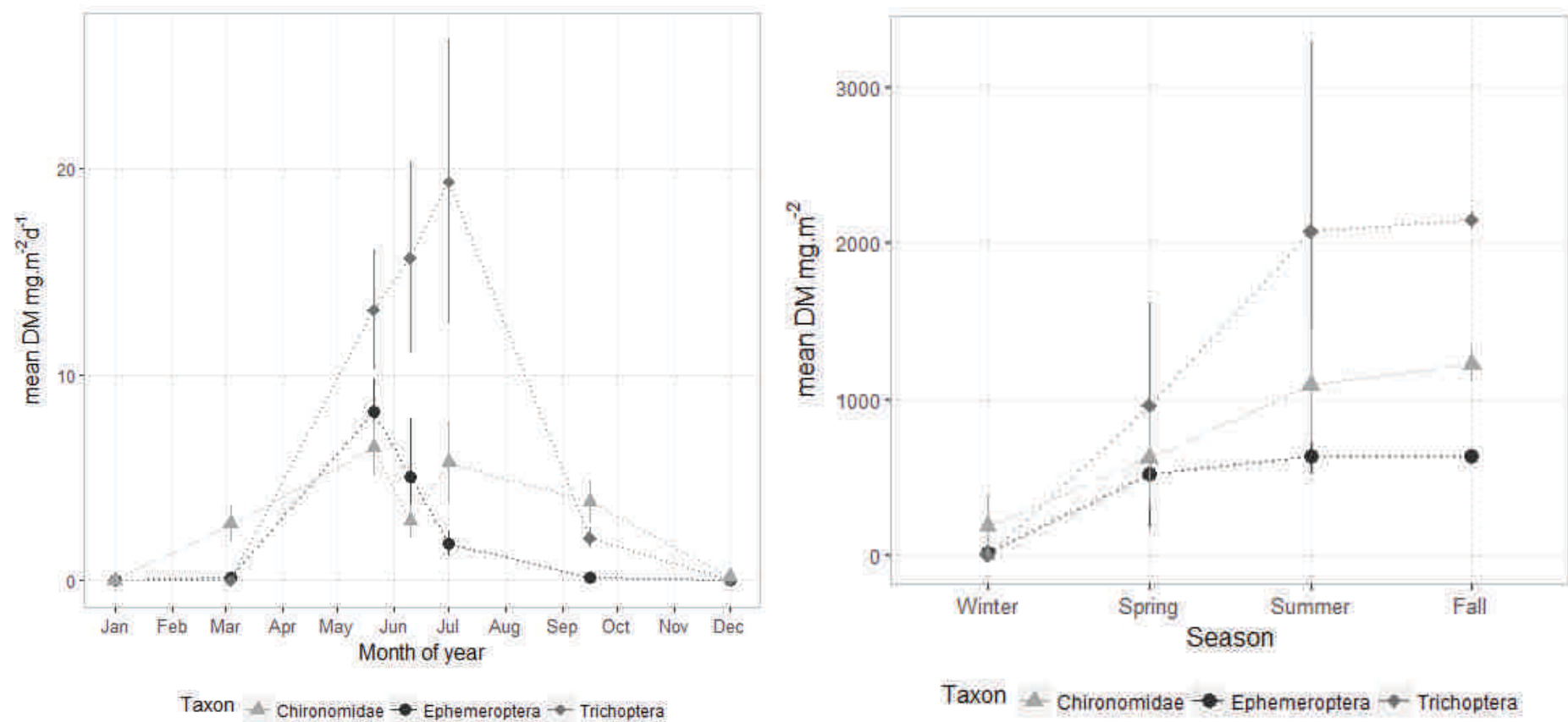


Figure 1. Emergence of aquatic insects dry mass (DM) all sites combined. Left panel shows mean ( $\pm 1$  standard deviation SD) daily values for each sampling date and taxonomic group. Right panel shows the mean ( $\pm 1$  SD) cumulative emergence by season (see text for details).

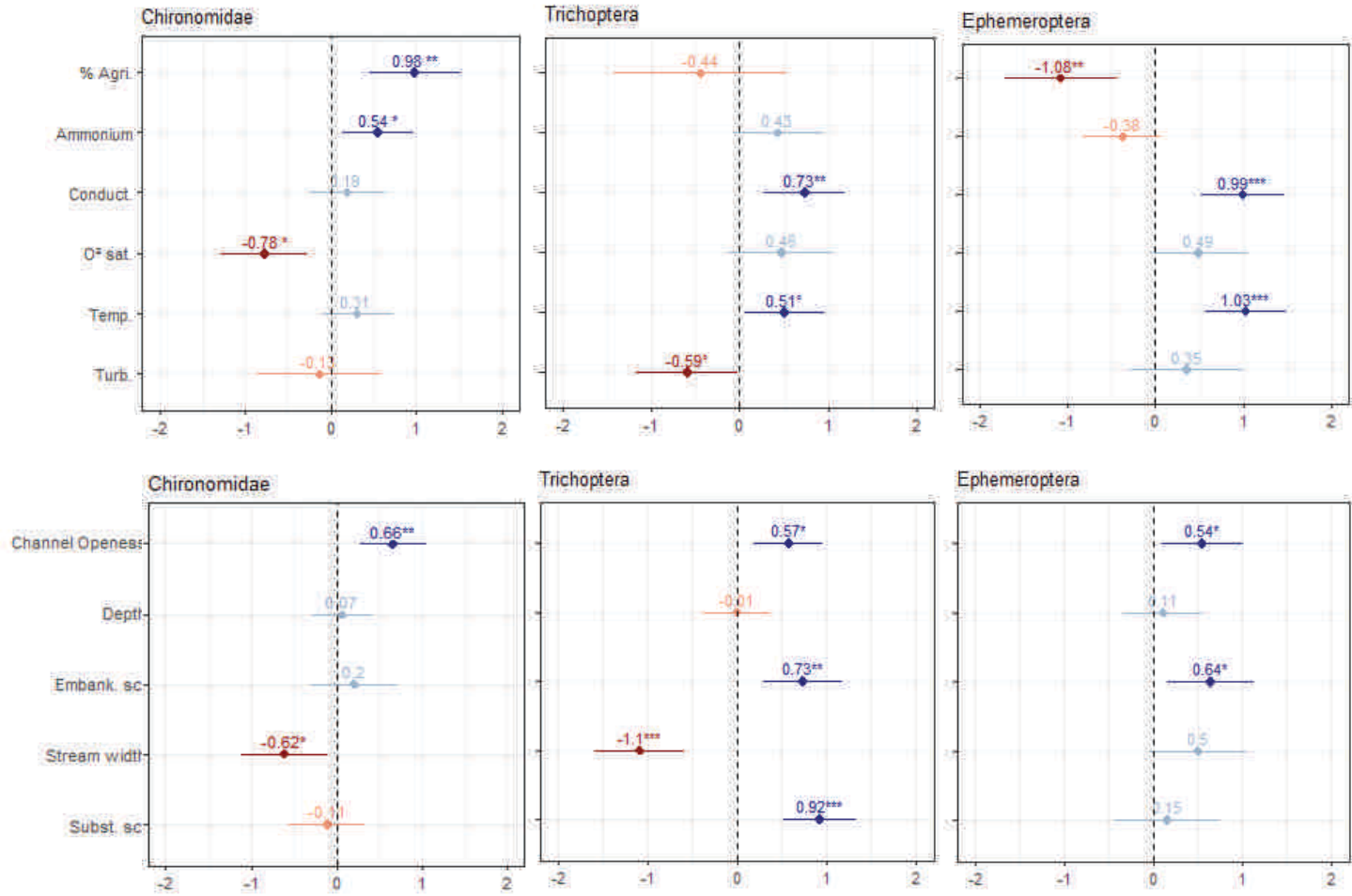


Figure 2. Standardized effect of environmental variables (mean effect size  $\pm$  90% confidence interval) at watershed and site scales on emerging biomass of Chironomidae (left), Trichoptera (middle) and Ephemeroptera (right). Upper panel: Watershed-scale variables; % Agri = Percentage of watershed area covered by agriculture (cropland and pasture), Ammonium = Ammonium (NH<sub>4</sub><sup>+</sup>) concentration in water, Conductivity =



Water conductivity, O<sup>2</sup> Sat. = Oxygen saturation, Temp. = Water temperature, Turb. = Water turbidity. Lower panel: Site-scale variables;

Channel Openness = Proportion of non-canopy cover over the stream channel, Depth = Water depth, Embank. Sc. = Stream width divided by bank height, Stream width = channel width, Subst. sc. = Substratum score, based on modified Wenworth classification (Cummins, 1962).

Colors indicate degree of positive or negative effect. ° =  $p < 0.10$ , \* =  $p < 0.05$ , \*\* =  $p < 0.01$ , \*\*\* =  $p < 0.001$

**Supplementary material for on-line publication only**

[Click here to download Supplementary material for on-line publication only: Raitif-et-al-v2.0\\_SuppMatr.docx](#)