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

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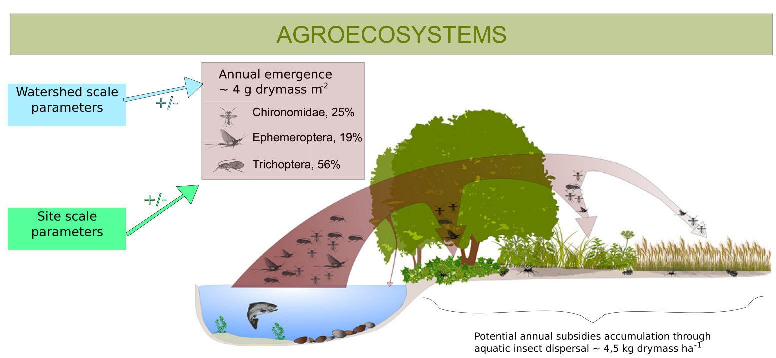
HIGHLIGHTS

- Aquatic insects emergence reaches 4 g drymass (DM) m^{-2} y^{-1} 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^{-1} y^{-1}.

GRAPHICAL ABSTRACT



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ABSTRACT

A growing amount of literature exists on reciprocal fluxes of matter and energy between ecosystems. Aquatic subsidies of winged aquatic insects can affect terrestrial ecosystems significantly, but this issue is rarely addressed in agroecosystems. By altering the production of benthic macroinvertebrates, agricultural practices could increase or decrease the strength of aquatic subsidies and subsequently the provision of several ecosystem services to agriculture. Effects of seasons and environmental variables on aquatic insect emergence were investigated in third-order agricultural streams in northwestern France. Most emerging dry mass (DM) of caught insects belonged to Trichoptera (56%), Chironomidae (25%) and Ephemeroptera (19%). We estimated that annual emerging dry mass of aquatic insects ranged between 1445 and 7374 mg/m^2/y depending on the stream. Seasonal variations were taxon-specific, with Ephemeroptera emerging only in spring, Trichoptera emerging in spring and early summer, and Chironomidae emerging throughout the year. The percentage of watershed area covered by agriculture, ammonium concentration and hypoxia positively influenced emerging DM of Chironomidae but negatively influenced Ephemeroptera. Emerging DM of Trichoptera and Ephemeroptera increased significantly as water conductivity and temperature increased. Channel openness increased the emerging DM of all taxonomic groups, but Chironomidae were more abundant in narrow, incised streams. Assuming that the biomass of aquatic invertebrates ultimately disperse toward terrestrial habitats, nutrient accumulations on land near streams were estimated to reach 0.5–2.3 kg C ha^{-1} y^{-1}, 0.1–0.5 kg N ha^{-1} y^{-1} and 0.005–0.03 kg P ha^{-1} y^{-1}, depending on the stream. This suggests a significant flux of aquatic nutrients to agroecosystems and the need for future studies of its potential influence on the ecosystem services provided to agriculture.

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1. Introduction

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

Adult aquatic winged insects are effective vectors of aquatic subsidies (Bartels et al., 2012) since they are abundant in almost all freshwater ecosystems, and their ability to disperse is substantial (Muehlbauer et al., 2014). Several studies have demonstrated their role as prey for terrestrial predators (*e.g.* birds, Gray, 1993; carabids, Hering and Plachter, 1997; spiders, Paetzold et al., 2005), providing a valuable source of nutrients for entire terrestrial ecosystems (Dreyer et al., 2015). Agricultural practices can regulate the production of benthic macroinvertebrates in streams by modifying water chemistry, eutrophication processes and pollution (Sallenave and Day, 1991; Liess and Von Der Ohe, 2005; Cross et al., 2006; Davis et al., 2010; Johnson et al., 2013a; Beketov et al., 2013); changing the flow regime and aquatic habitats (Rabení et al., 2005; Kennedy and Turner, 2011; Wagenhoff et al., 2011; Magbanua et al., 2016); and modifying riparian vegetation (Deegan and Ganf, 2008) and water temperature (Nagasaka and Nakamura, 1999; Sponseller et al., 2001). Consequently, the influence of agriculture in watersheds on the emergence of stream winged insects could increase or decrease the strength of aquatic subsidies provided to terrestrial ecosystems. For instance, stream insect communities are usually dominated by small taxa with greater ability to disperse by flight in agricultural landscapes (Stenroth et al., 2015; Carlson et al., 2016; Greenwood and Booker, 2016). While insect emergence depends on season and weather conditions (Corbet, 1964), few studies quantify these influences or analyze their variability in agricultural landscapes. For instance, Shieh et al. (2003) and Gücker et al. (2011) published seasonal variations of aquatic insect production in two agricultural streams, but not emergence data. However, information on spatial and temporal variation in the emerging biomass of aquatic insect taxa is essential to accurately estimate the amount of aquatic subsidies annually produced by agricultural streams.

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

2. Methods

2.1. Study sites

The study was performed in the Ille-et-Vilaine county (Brittany), western France. This area sits on sedimentary rocks (schist and sandstone) with occasional layers of aeolian loam deposits. The climate is oceanic, with a mean annual temperature of 10.5–12.5 °C and cumulative annual rainfall of approximately 700 mm (peaking in fall and winter) (Météo France, 2017). Mean annual discharge of third-order streams in this part of Brittany is 0.65 $m^3\ s^{-1}$, ranging from 0.06 $m^3\ s^{-1}$ in September (end of summer) to 2.23 $m^3\ s^{-1}$ in March (end of winter).

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

2.2. Aquatic insect sampling

We estimated aquatic insect emergence by deploying two emergence traps at each site. The traps consisted of a floating pyramidal tent (1 m^2 at its base) made of nylon mesh and anchored to shorelines with ropes. Because substratum and water velocity strongly affect aquatic insect communities (Tachet et al., 2010), one trap was set in a deep and silty habitat and the other in a shallow gravel-pebble habitat to reflect the natural habitat heterogeneity of each site. Adult insects were collected in a plastic bottle placed at the top of the trap and filled with a mix of water, concentrated detergent and propylene glycol (approx. 20%) to preserve insects. Upon collection, aquatic insects were stored in alcohol (70%).

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

2.3. Emergence abundance, dry mass and secondary production

A stereomicroscope was used (1) to identify adult aquatic insects to the order level for Ephemeroptera, Plecoptera, and Trichoptera, and (2) to separate Chironomidae from other Diptera, the most abundant taxon in our sampling. Other insects, mainly Diptera, were not considered because their contribution to total DM was low and did not justify the time-consuming work required for identification. For Chironomidae, sub-samples (1/2, 1/4 or 1/8) were obtained with a Motoda splitter (Motoda, 1959) when abundance was too high. Insects were