IRRIGATION DITCHES AS NOVEL INTERMITTENT STREAM NETWORKS THAT PROVIDE ENERGETIC SUBSIDIES TO TERRESTRIAL ECOSYSTEMS VIA AQUATIC INSECT EMERGENCE

by

Nathaniel Maxwell Heili

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ABSTRACT

Non-perennial streams dominate fluvial networks, comprising over half of lotic ecosystems globally. Although little attention has been paid to these systems relative to those that are perennial, perhaps even less is known about the ecology of artificial intermittent streams. In irrigated river valleys, ditches comprise a substantial proportion of surface water networks, but little is known about their relative contributions to lotic habitat, freshwater biodiversity, and ecosystem processes. Because ditches are abundant and permeate arid floodplains, they create new opportunities for cross-ecosystem subsidies through emergence of adult aquatic insects. Here, we mapped the extent of an irrigation ditch network relative to natural surface waters in the Gallatin River Valley, Montana, USA. We also quantified the magnitude, composition, and phenology of aquatic insect emergence in ditches throughout a full irrigation season and compare emergence to nearby natural streams in the valley. We found that non-perennial streams, both natural and artificial, dominated the surface-water network, representing over 70% of total length. Irrigation ditches also constituted 37% and 23% of total length and surface area of all waterbodies in the valley, respectively. Insect emergence production from ditches averaged 32.0 mg m^{-2} d^{-1} and exceeded fluxes from nearby natural streams, which contained more classically sensitive taxa (i.e., Ephemeroptera, Trichoptera, and Plecoptera). Ditches varied in water temperature, substrate size, and flow permanence, leading to distinct emergence timing, species composition, and magnitudes of biomass flux. One of the study ditches dried early because of more 'junior' water rights, but this ditch provided the largest emergence subsidy, including a pulse of dipteran emergence at the onset of flow cessation. Annual production from intermittent ditches was ~6 g m^{-2} y^{-1} and was comparable to estimates from natural perennial streams. Although our study focused on one river valley of the western US, results suggest that these ecosystems contribute broadly to lotic habitat, heterogeneity, and cross-ecosystem subsidies via aquatic insect emergence. Given increasing demand for water and changes to flow regimes and water management driven by climate change, additional study is needed on these novel and underappreciated artificial ecosystems.

INTRODUCTION

Non-perennial streams dominate river networks, comprising more than half of lotic ecosystems globally (Messager et al. 2021). In recent decades, many studies have documented the unique contribution of non-perennial streams to regional biodiversity and ecosystem function (Acuña et al. 2014, Datry et al. 2017). Although little attention has been paid to these systems relative to those that are perennial, perhaps even less is known about *artificial* non-perennial streams constructed through human activities (*sensu* Clifford and Heffernan 2018, Datry et al. 2023). Because humans engineer and appropriate water resources at a global scale, their activities have altered aquatic habitat to an extent that rivals or exceeds natural fluvial processes. In many river networks, artificial streams may comprise a substantial total of lotic habitat, and although they are highly modified, recent studies suggest that these ecosystems can harbor abundant and unique biota (Herzon and Helenius 2008, Chester and Robson 2013, Lin et al. 2020, Huttunen et al. 2024). Yet, despite their ubiquity and large potential contribution to community- and ecosystem-level processes, the ecology of artificial streams remains vastly understudied, especially relative to non-perennial streams.

Agricultural ditches and canals, typically used to drain or deliver water for crop production, are among the most extensive artificial stream ecosystems globally (Clifford and Heffernan 2018, Peacock et al. 2021, Tank et al. 2021). Yet, from both a cartographic and an ecological perspective, our knowledge of ditch networks is limited because of their low perceived value and because they are typically unrepresented in hydrological data sets (Lidberg et al. 2023, Robb et al. 2023). In many regions, ditch networks are extensive, and their total length exceeds that of natural streams. For instance, agricultural ditches account for over 50% of

all surface waters in the Netherlands (Highler 1989), Sweden (Paul et al. 2023), and Great Britain (Brown et al. 2006), and recent estimates suggest that ditches comprise more than twice the surface area of all natural streams in the United States (Clifford and Heffernan 2018). In recent years, growing concerns about water scarcity and shifting land use have directed focus towards understanding the potential influence of ditches, both positive and negative, on aquatic habitat and hydrologic processes.

Agricultural ditches often provide services beyond their intended purpose, contributing to stream habitat and fluvial processes (Dollinger et al. 2015, Biggs et al. 2016, Tank et al. 2021, Rideout et al. 2021). Unlined ditches recharge underlying aquifers (Fernald and Guldan 2007, Michalek and Sutherland 2020, Ketchum et al. 2023), distribute erosive forces during high flows (De Laney 1995, Powell et al. 2007), retain nutrients and sediments (Davis et al. 2015, Mahl et al. 2015), and influence greenhouse gas exchange (Peacock et al. 2021, Wu et al. 2023). Ditches also harbor diverse aquatic biota (Williams et al. 2004, Stammler et al. 2008, Hill et al. 2016, Clifford and Heffernan 2023) and provide habitat for riparian plant and animal species in an otherwise homogenous landscape (Mauritzen et al. 1999, Mazerolle 2005, Heroldová et al. 2007, Herzon and O'Hara 2007, Carlson et al. 2019). Yet, despite recent awareness of these factors, there is still relatively little known about the ecological role of ditches in fluvial landscapes (Koetsier and McCauley 2015, Meier et al. 2017, Gething and Little 2019), which may be particularly important given the transition of agricultural lands to developed lands in many regions (Hansen et al. 2002, Baker et al. 2014).

One important way that ditch networks may influence the ecology of fluvial and terrestrial landscapes is by increasing opportunities for energy and material transfer between

surface water and the terrestrial environment (i.e., resource subsidy; Polis et al. 1997, 2004). In agricultural landscapes, where river-floodplain interactions have been historically severed, these transfers may be particularly important. It is well known that emerging aquatic insects provide an important resource subsidy to recipient terrestrial communities (Nakano and Murakami 2001, Power et al. 2004, Baxter et al. 2005, Twining et al. 2018) and detrital food webs (Hoekman et al. 2011, Bartrons et al. 2013, Dreyer et al. 2015, Wesner et al. 2019). Because ditches spread laterally onto floodplains, they extend the 'signature' of the river onto land, and likely enhance importance of aquatic insect emergence at large landscape scales (Gratton and Vander Zanden 2008, Muehlbauer et al. 2014). Characterizing the magnitude of energy subsidy provided by irrigation ditch networks may reveal a strong influence on riparian and terrestrial ecosystems that have been altered by extensive artificial irrigation or drainage systems.

Ditches may also influence fluvial landscapes by increasing heterogeneity in lotic habitat (Loreau et al. 2003, Datry et al. 2017), especially at locations within ditches that are distant from natural sources. Increased spatial and temporal variation in lotic environmental conditions (i.e., flow, water temperature, substrate) could alter aspects of community composition (Bogan and Lytle 2007, Tonkin et al. 2017) and enhance regional beta-diversity (Datry et al. 2014, 2016). With respect to insect emergence, the novel mosaic of environmental conditions created by ditches (especially water temperature and flow variability) could generate emergence patterns that are unique in their phenology, duration, and magnitude (Whiles and Goldowitz 2001, Uno 2016, Anderson et al. 2019, Adams et al. 2023). In other words, ditch networks likely introduce novel heterogeneity in emergence that diversifies the 'portfolio' of aquatic subsidies at the river network scale (Schindler and Smits 2017).

The goal of this research was to build understanding of the energy subsidies within a river network heavily engineered by canals and ditches that intermittently deliver surface water for irrigation. To achieve this goal, we quantified the magnitude, composition, and phenology of aquatic insect emergence across the irrigation ditch network in the Gallatin River Valley in southwestern Montana, USA. We also compared our biomass flux estimates to nearby natural streams and estimates from other studies. In addition, we quantified the spatial extent and surface area of the ditch network relative to natural perennial and intermittent streams, and described the network's distributary-like topology. Based on previous studies in agriculturally influenced streams, we predicted that irrigation ditches would exhibit lower taxonomic richness but higher daily biomass flux of emerging insects than natural perennial streams. In addition, we predicted that ditches that dry more frequently and for longer duration (due to junior water rights and reduced connectivity to rivers) would be less diverse and productive per unit area than those that stay wetted throughout the irrigation season. Finally, we developed scenarios to address how climate-induced changes to flow and water use might affect total biomass export from the Gallatin Valley ditch network. Our results demonstrate that irrigation ditch networks create novel lotic ecosystems that likely play a substantial role in driving energetic subsidies across agricultural regions of the western US.

METHODS

Study Area

Our study was conducted in the Gallatin River Valley in southwestern Montana, one of the oldest and most productive agricultural regions in the state. Precipitation in the valley is insufficient to support optimal crop production throughout the growing season, and surface water is therefore diverted from natural streams and distributed to farms through an extensive, high density network of unlined ditches (Buck and Bille 1953, Lonsdale et al. 2020, Michalek and Sutherland 2020; Figure 1). The history of irrigation in the Gallatin Valley dates back to the mid-19th century, and today over ninety primary ditches (i.e., connected to a natural stream or river) and abundant smaller lateral ditches (i.e., connected to a primary ditch) convey water across the landscape to farms. The primary source of surface water for irrigation is the West Gallatin River, although water is diverted from numerous other perennial and intermittent streams, including the East Gallatin River and multiple smaller tributaries.

The irrigation season in the Gallatin Valley typically spans from May to early November, although the timing fluctuates to some degree based on snowpack, weather, and crop water requirements. The start date of water diversion is determined by each respective ditch company (i.e., the entity responsible for the management, maintenance, and regulation of the ditch). Water diversion is mainly controlled by manual adjustment of headgates, though water can be redirected into a ditch at an intersection with a stream. During the first part of the irrigation season (May-June), discharge in the West Gallatin River is at its peak due to snowmelt, and irrigators often take advantage of high flows to remove debris from ditches. Surface water is typically diverted throughout the growing season until water shortages occur among users (i.e.,

July-September), at which point the regional water commissioner allocates water resources based on water rights or 'seniority' (i.e., date of the original water right). Towards the end of the irrigation season (October-November), water may also be diverted into previously dry ditches and distributed onto farm fields to enhance water stored in the soil for the following growing season. During our study, water was conveyed through ditches until early November when air temperature dropped below freezing and ice developed in stagnant pools. Together, these dynamics highlight a diverse pattern of flow permanence across the ditch network that is driven by both natural and anthropogenic factors.

Mapping And Geospatial Analysis

We used existing waterway shapefiles to categorize and quantify the total length and area of the surface-water network in the Gallatin River Valley. These data were sourced from various entities including Gallatin County GIS Department, Montana State University, Gallatin Conservation District, and unpublished data from Montana Bureau of Mines and Geology. We defined the spatial extent of the valley as the area within the lower Gallatin Watershed boundary designated by the Montana Department of Environmental Quality (Montana DEQ 2013). High relief mountainous regions in the headwaters of the watershed are not extensively utilized for agriculture and were thus excluded from this boundary, leading to a total area of 1588 km^2. We used ESRI aerial imagery available in ArcPro v3.2 (ESRI 2024) to add or remove visually evident or absent waterbodies from the aggregated data, as appropriate. Although the categorization of waterways (perennial, non-perennial, and ditch) was predetermined within the existing shapefiles, some of these were mislabeled. We therefore categorized irrigation ditches as artificial (human-constructed) non-perennial waterways that are used for irrigation. We

acknowledge that some of these ditches may represent historic stream channels, but they have been altered to such an extent that they now only serve to deliver water. We then quantified the total length of ditches, perennial streams, and natural non-perennial streams, using the "Summarize Within" geoprocessing tool (ESRI 2024).

We studied four 'focal' primary irrigation ditches within the network to sample habitat and insect emergence: Beck Border Ditch, Farmer's Canal, Lower Middle Creek Supply, and Lowline Canal (Figure 1). We chose these ditches because they broadly represent existing variation in water rights and abiotic factors across the ditch network (Table S1). In each focal ditch, ten sampling sites were located with increasing distance from a natural tributary or mainstem river. We determined the total length of each ditch and the distance of each sampling site from a natural river using ArcPro (ESRI 2024). Notably, distance from a natural river was not always based on distance from a headgate (i.e., point of water diversion), but also represented the distance from perennial stream crossings within a given ditch.

Quantifying the surface area of each waterway type required estimates of wetted width across the valley. We therefore measured wetted width to the nearest 0.1 m using a tape measure at each sampling site (n = 30) from June to November in 2022, leading to a total of 206 observations. We also measured wetted width in five perennial streams of varying size classes (46 observations) from June to August 2023. We supplemented these measurements with remote observations using Google Earth Pro, providing an additional 58 observations of unsampled perennial streams and 34 observations of non-perennial streams. For each waterway type, we sampled from a Gaussian distribution around the mean defined by the standard deviation of measured widths and multiplied by the respective value of total length, generating an ensemble

of 1000 estimates of surface area. We conducted a similar but separate analysis for the two major perennial streams in the valley (East and West Gallatin Rivers) because they were substantially wider than other perennial streams.

During our analysis, we recognized the likelihood of overlooking the many smaller secondary and tertiary ditches that deliver water to individual farms or farm clusters. To address this potential bias, we used the "Create Fishnet" geoprocessing tool (ESRI 2024) to create 5 km^2 grids across the valley. We conducted a detailed visual assessment of fifty randomly selected grids by digitizing ditches that were previously omitted in our original map products. We summarized the mean areal density of omitted ditches across the fifty 5 km^2 grids and multiplied this value by the total surface area of the valley to produce a revised estimate of ditch network length. In addition, we carefully mapped the distributary configuration (i.e., primary, secondary, and tertiary ditches) of one major ditch within the network and quantified the length of each branch using the "Summarize Within" geoprocessing tool (ESRI 2024).

Aquatic Insect Emergence

We sampled emerging aquatic insects continuously at each sampling site from 7 June to 11 October 2022. At each site, we deployed three replicate floating emergence traps (Cadmus et al. 2016) to passively collect adult aquatic insects from a known surface area (0.33 m^2). Each floating trap was attached to a piece of rebar anchored to the stream bed. Trap walls consisted of 500 μm mesh netting and a collection bottle with sufficient surface area for insects to colonize upon emergence. The collection bottle contained 75% ethanol which immediately preserved emerging insects. Collection bottles were sampled on 4-7 day intervals following established protocols (Malison et al. 2010, Cadmus et al. 2016).

In an effort to compare emergence flux from ditches to natural streams in the valley, we sampled insect emergence in five perennial streams from 7 June to 1 August 2023 using the same methods described above. Although the duration of this sampling was shorter than the previous summer, we focused on the time of the year that would allow comparisons of peak production. For this effort, we quantified emergence at one site on each of the five perennial streams with three replicate emergence traps at each site.

In the laboratory, individual samples were poured into a shallow petri dish and manually distributed to avoid overlap of individuals. We then used a Sony Alpha 7S full-frame camera with a Zeiss Distagon wide-angle lens and an enclosed container with fluorescent lights to capture a high-resolution image of all individuals in the petri dish. Samples with particularly large numbers of individuals were subsampled with a Folsom Plankton splitter prior to imaging (Van Guelpen et al. 1982), typically to one-half of the total sample. All individuals were identified to the order level, enumerated, and measured lengthwise to the closest thousandth of a millimeter using the software Image-J (Schneider et al. 2012). If a sample had more than twenty individuals of the same order, we measured twenty individuals and assumed a similar length distribution for the remainder. We did not enumerate or measure non-adult aquatic insects that were found in collection bottles (e.g., bees, ants, spiders).

To estimate individual biomass in dry mass (DM) of emerging insect, we used published order-level length-mass regressions (Sabo et al. 2002). Individual biomass values were multiplied by the total abundance of a given taxon to estimate taxon-specific emergence dry mass for the sampling period. We removed Odonata from the analysis because of low numbers and disproportionately high biomass. To make comparisons among and within ditches, biomass

estimates were converted to daily emergence flux (mg DM m^{-2} d^{-1}) by dividing biomass by the trap area (0.33 m^2) and the number of days between sampling events.

Physical Factors

Water temperatures were recorded at 1-hour intervals from 2 June to 8 November 2022 at each sampling site using HOBO Pendant water temperature loggers (Onset Data Loggers, Bourne, Massachusetts, USA) secured to a fence post and shaded with a UV-safe PVC tube. We defined the wetted period as 7 June to 28 July when making statistical comparison among ditches, but we continued to measure temperature outside of this period to understand the influence of drying and freezing on aquatic insect emergence. We determined the size of benthic substrate size by taking a digital image of the bottom during dry periods at each sampling site. We then qualitatively categorized substrate into gravel (>= 2mm) or sand (<= 2mm) by measuring substrate size within three one square meter quadrats using Image-J.

Analysis

Comparing Emergence Flux from Ditches and Streams

To examine whether emergence flux magnitude differed between ditches and streams, we used a linear mixed-effects model, where natural log of emergence flux (mg DM m^{-2} d^{-1}) was the response variable, Julian day and waterway type were fixed effects, and location (i.e., stream name or focal ditch) and individual emergence trap were nested random effects. We log transformed emergence flux because it was positively skewed and not normally distributed. We assumed a lack of confidence in difference between emergence in ditches and streams if the

confidence intervals for the waterway type coefficient overlapped zero and the p-value was greater than the commonly used threshold of 0.05.

Predicting Emergence Flux from Ditches

We used a Bayesian generalized additive mixed model (GAMM) via R package *brms* (Bürkner 2017) to estimate emergence flux (mg DM m^{-2} day^{-1}) for each day of sampling from ditches (7 June to 11 October) following the approach of Wesner et al. (2019). In this case, daily emergence flux was the response variable, Julian day was the smoothed fixed effect, and focal ditch, site, and individual emergence traps were nested random effects. We chose to apply a GAMM because emergence patterns were highly nonlinear over this time period and the smooth fixed effect of time is fitted with a spline function (Bürkner 2017, Wood 2017). We used a weakly informative prior for the intercept parameter based on a Gaussian distribution from 32 estimates of emergence flux from lotic ecosystems (Table S2). All other parameters contained vague priors based on standard probability distributions (e.g., gamma, Cauchy, Student's t). Graphical comparisons of the prior and posterior distributions indicated that the influence of our priors on the outcome was weak relative to the influence of the data (Figure S2).

After fitting the model, we simulated emergence flux from the posterior distribution using the *fitted* function from *brms*. Posterior distributions were sampled over 1000 iterations in each of 4 Markov chains. The resulting ensemble of 4000 estimates of daily emergence flux (mg DM m^{-2} d^{-1}) characterized the inferential uncertainty for each of the 126 days between 7 June and 11 October. We report the median value of emergence flux and the 95% quantiles of the ensemble as the credible interval (95% CrI).

We used another GAMM analysis to test the degree to which the magnitude and phenology of emergence biomass flux differed among ditches during our sampling period. We used the same model structure as described above, but we allowed the smoothed predictor of Julian day to vary by ditch which resulted in posterior estimates of daily biomass flux between 7 June and 11 October for each focal ditch. We also calculated the ensemble of cumulative emergence production estimates (g DM m^{-2}) from 7 June to 11 October for each study ditch to examine differences in the seasonality of biomass accumulation.

For all Bayesian models, the posterior distribution was sampled using the Hamiltonian Monte Carlo algorithm in *rstan* (Stan Development Team 2024) via the *brms* package in R (*v*4.3.2, R Core Team 2023). We ran 4 chains with 2000 iterations each, with the first 1000 discarded as a warmup.

Physical Conditions and Community Composition of Emergence Along Ditches

To address the effect of distance from a natural stream or stream crossing on the community composition of emergence, we used a beta regression model, with the proportion of total emergence comprised of Ephemeroptera, Plecoptera, and Trichoptera as the response variable and distance from a natural stream or stream crossing as the fixed effect via the *betareg* package (Cribari-Neto and Zeileis 2010) in R.

We compared depth, wetted width, and water temperature among ditches using analysis of variance, where the factor of interest was the response variable and study ditch as the predictor. We used linear regression to examine how these factors differed with distance from a natural stream, where the factor of interest was the response and distance from a natural stream

as the predictor. Finally, we used logistic regression to determine if our two categories of substrate size (gravel and sand) varied with distance from a river.

Valley-Wide Estimate of Emergence Flux and Potential Effects of Climate Change

We combined estimates of emergence flux from ditches and streams with estimates of the areal extent of these habitats to produce estimates of emergence flux from all lotic water bodies in the valley during irrigation season (15 May to 1 November). We excluded intermittent streams in the valley from this analysis because they typically run dry prior to the onset of the irrigation season (*pers. obs.*).

Given that our observations of emergence flux from natural streams and ditches were comparable (see *Results*), we assumed that median daily emergence flux from natural streams was equivalent to ditches. We therefore used the model fit to data from ditches to estimate the integrated biomass export (kg DM y^{-1}) from both the ditch network and perennial streams. To estimate emergence for days before and after our sampling (n = 44 days if we assume a typical irrigation season lasts from 15 May to 1 November; Buck and Bille 1953), we applied the median daily emergence value from the posterior distribution on the first and last day of sampling (see *Supplementary*). For the time during our sampling period (7 June to 11 October), we used the *predict* function from *brms* to generate an ensemble of daily emergence estimates. These predictions had wide intervals because they incorporate uncertainty by using the standard deviation of the random effects term (Bürkner 2017, Wesner et al. 2019). We estimated valley-wide areal flux (g DM m^{-2} y^{-1}) by summing the ensembles of daily emergence flux between 15 May and 1 November, which resulted in a posterior predictive distribution of annual emergence

flux. We then multiplied annual emergence production by the aerial extent (m^2) of each habitat to estimate total export from components of the surface-water network.

We addressed the effects of climate-induced changes to stream flow on aquatic insect emergence subsidies from the ditch network by simulating three potential irrigation seasons that varied in their timing, duration, and extent: a wet year (longer irrigation season; ditches wetted from 15 April to 1 November), a dry year (ditches with senior water users wetted from 15 May to 1 November, those with more junior water users wetted from 15 May to 1 July), and an extreme drought year (shortened irrigation season in all ditches; wetted from 15 May to 1 July). Under the dry year scenario, we divided the network extent equally between junior and senior users. We then applied the median posterior estimate of emergence for each day of the simulated irrigation season and multiplied by the wetted area of the network to estimate total annual biomass export (kg DM y^{-1}) for each respective scenario.

RESULTS

Surface-Water Network Characteristics

Intermittent streams, both natural and artificial, dominated the surface-water network of the Gallatin River Valley, constituting over 70% or 2284 km of total network length (Figure 1). Irrigation ditches represented the longest waterbody type in the valley at 1152 km, only slightly longer than natural intermittent streams (Figure 2). Wetted surface area of the ditch network was estimated at 5.8 ± 2.1 km^2 (mean ± standard deviation), which was the second largest area for a given lotic habitat type (Figure 2). Overall, irrigation ditches were an integral component of the valley surface-water network, representing 37% and 23% of the total length and area, respectively. Detailed mapping of a section of Farmers Canal resembled a distributary-like network that bifurcated multiple times after the point of primary diversion (Figure 1). Within this section, lateral (i.e., non-primary) segments were the most abundant and constituted over 65% of total length (Figure S1), demonstrating the role of secondary branches to surface water delivery.

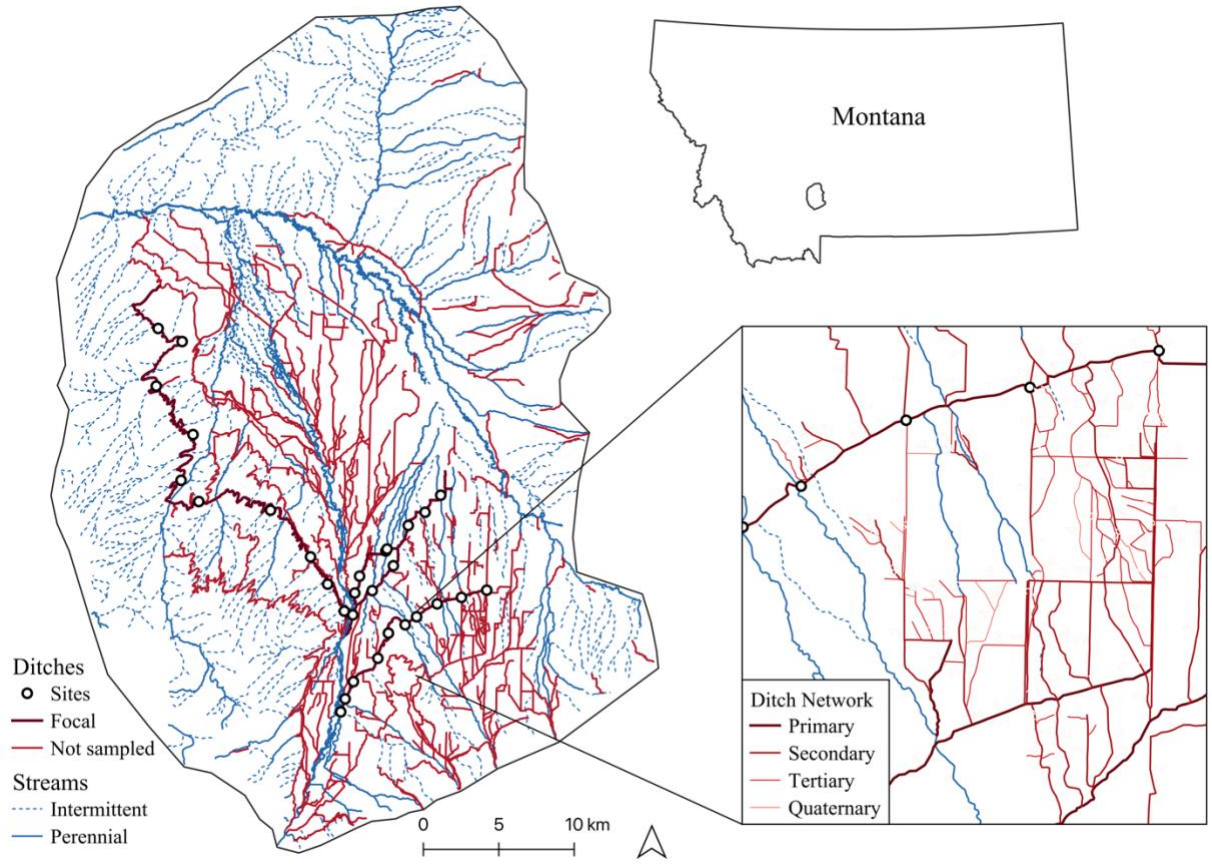

Figure 1. Surface-water network in the Gallatin River Valley, Montana, USA. On the left map, blue lines represent perennial streams, dashed lines represent intermittent streams, red lines represent irrigation ditches, and the black lines represent the focal ditches with circles showing the sampling site locations. The inset shows a major section of a ditch network, highlighting its distributary character with multiple branches (e.g., primary, secondary, tertiary, quaternary).

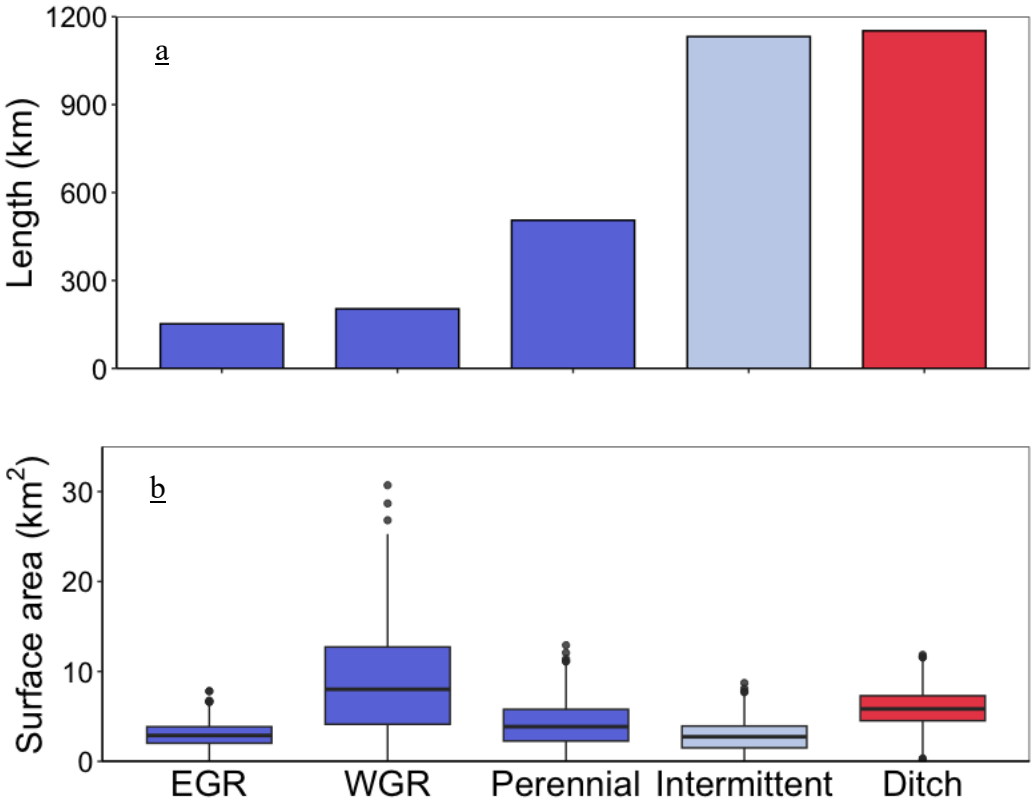

Figure 2. The estimated (a) length (km) and (b) area (km^2) of different categories of the surface-water network in the Gallatin River Valley, Montana. Surface area was estimated using variation in wetted width, and the box and whisker distributions represent the median (central line), interquartile range (edges of box), 2.5^{th} and 97.5^{th} percentiles (ends of whiskers), and values outside of this range (points). The dark blue represents perennial streams, light blue represents natural intermittent streams, and red represents irrigation ditches. We preformed this analysis separately for the two largest perennial streams in the valley, the East Gallatin River (EGR) and the West Gallatin River (WGR).

Water temperature varied substantially in the study ditches throughout the irrigation season and spatially across the valley. Temperatures were the coolest at the beginning of the irrigation season, peaked in mid-August, and then declined through October (Figure 3). Ditches that divert surface water further downstream in the West Gallatin River were warmer, reflecting the longitudinal temperature gradient of the West Gallatin River. Water temperatures in the study ditches also increased with distance from a natural river or stream crossing (linear regression, r^2

= 0.1, $p < 0.001$, Figure 4a), but this relationship was weakened by perennial stream crossings where mixing occurred. Lowline Canal extended over 60 km without interacting with other waterways (Table S1) and thus had the most pronounced longitudinal temperature gradient. In this ditch, temperatures at the furthest downstream site were ~4.5 °C warmer than those at the headgate. Benthic substrate size also transitioned from gravel to sand as distance from a river or stream crossing increased, though confidence in a logistic slope differing from zero was not significant (logistic regression, $p = 0.2$, Figure 4b).

Flow permanence during the irrigation season depended on water rights in study ditches (Table S1). Three of the four study ditches were managed for irrigators with relatively old (i.e., 'senior' or established before 1890) water rights and consequently contained water throughout the entire irrigation season. In contrast, the Lowline Canal was established more recently (i.e., 1924), and flows in this ditch were periodically halted during water shortages (August-September). Diversion interruptions to junior water rights resulted in about 40 additional days without flow compared to the other study ditches, occurring during the middle of the irrigation season. However, low-lying areas within the canal formed pools during flow cessation that remained wetted for some weeks after the headgate was closed and flow ceased (see Figure 3c for illustration). Flow resumed in the Lowline Canal after water was distributed to canals with senior rights, rewetting previously dry substrates.

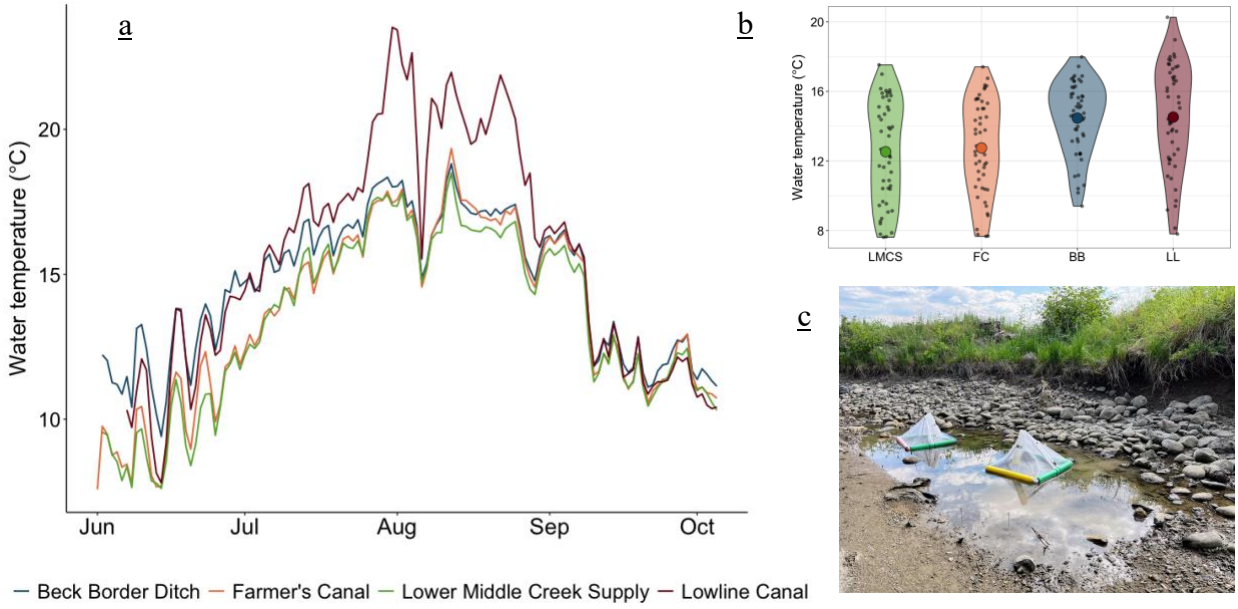

Figure 3. (a) Average water temperature (°C) for each study ditch plotted over time during the irrigation season. (b) Distributions of water temperature in each study ditch during the wetted period (7 June to 28 July). Colored points on the violin plot represent mean temperature during the wetted period. Lowline Canal was the only study ditch that dried during the irrigation season, and disproportionately higher temperatures in late-July and August reflected ambient air temperature. (c) Some low-lying segments within Lowline Canal remained wetted for weeks after the headgate closed, and water was rediverted occasionally during September. Ditch names are abbreviated, where LMCS is Lower Middle Creek Supply, FC is Farmer's Canal, BB is Beck Border Ditch, and LL is Lowline Canal.

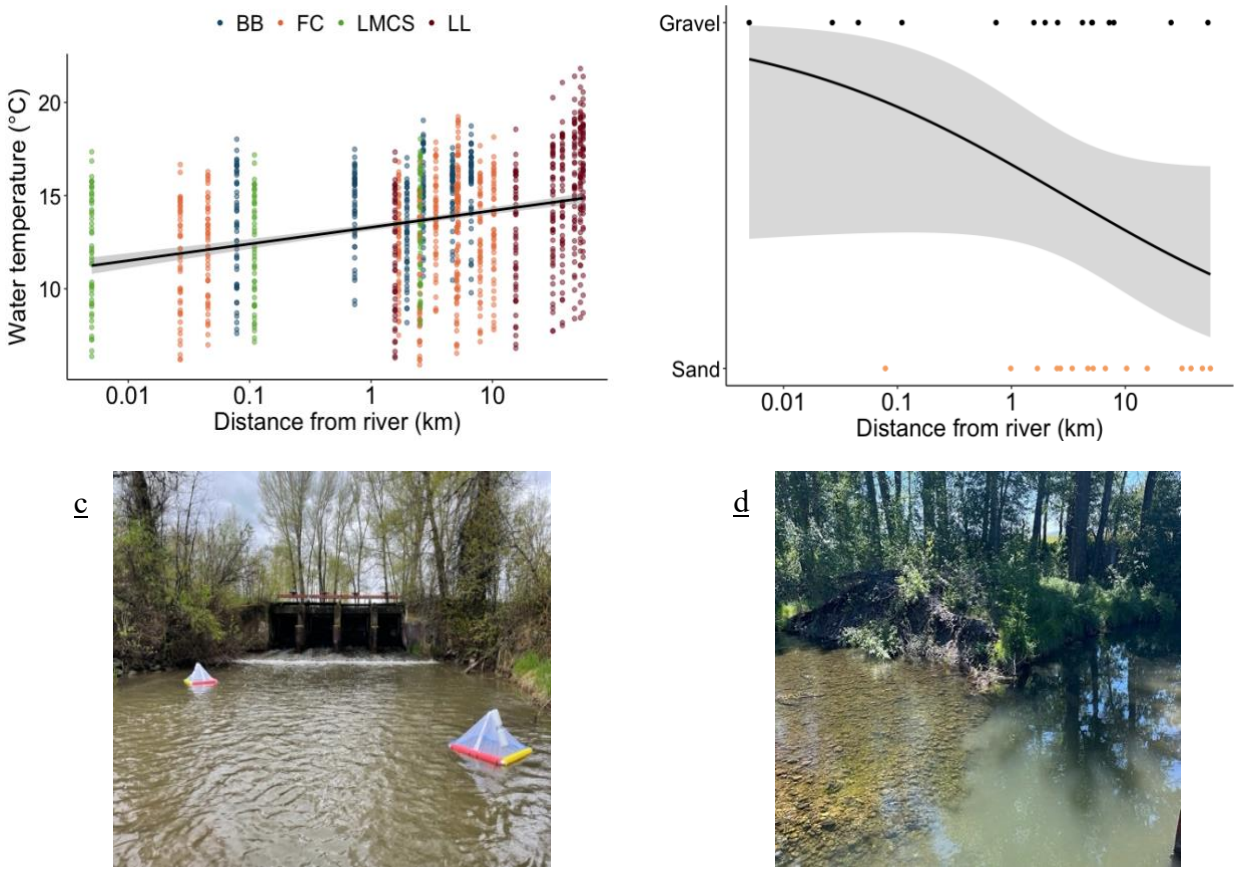

Figure 4. Changes in (a) average water temperature (°C) and (b) substrate type within ditches as a function of distance from a river or stream crossing (km). The dark line is the fitted regression line from (a) linear regression ($p < 0.001$) and (b) logistic regression ($p = 0.2$). The light gray area is the 95% confidence interval. Study ditch names are abbreviated as in Figure 3. Distance from river was determined from (c) the point of water diversion or (d) a perennial stream crossing (stream is left fork, ditch is the right fork).

Aquatic Insect Emergence

Comparing Emergence Flux from Ditches and Streams

Our data showed little evidence that the magnitude and phenology of aquatic insect emergence (mg DM m^{-2} d^{-1}) differed between irrigation ditches and streams in the Gallatin River Valley, MT (linear mixed effects model; $p = 0.55$), but the community composition of emerging insects appeared to vary strongly (Figure 5). In natural streams, over 60% of total biomass export

was attributed to Ephemeroptera, Plecoptera, and Trichoptera (EPT), whereas these orders only accounted for 24% of emergence biomass in ditches. The mean biomass emerging from ditches was modestly higher than that of streams (on average 14.0 mg DM m^{-2} d^{-1} greater), driven by large numbers of emerging Diptera (predominately Chironomidae; > 3000 ind. m^{-2} d^{-1} in some locations).

Median predicted annual emergence flux from ditches was 5.9 g DM m^{-2} y^{-1}, and the credible intervals fell within the range of 32 published values in perennial streams (Figure 6; Table S2). Median emergence from perennial streams was about 1.0 g DM m^{-2} y^{-1} less than that of intermittent ditches, demonstrating elevated daily production sustained throughout the irrigation season (Figure 5).

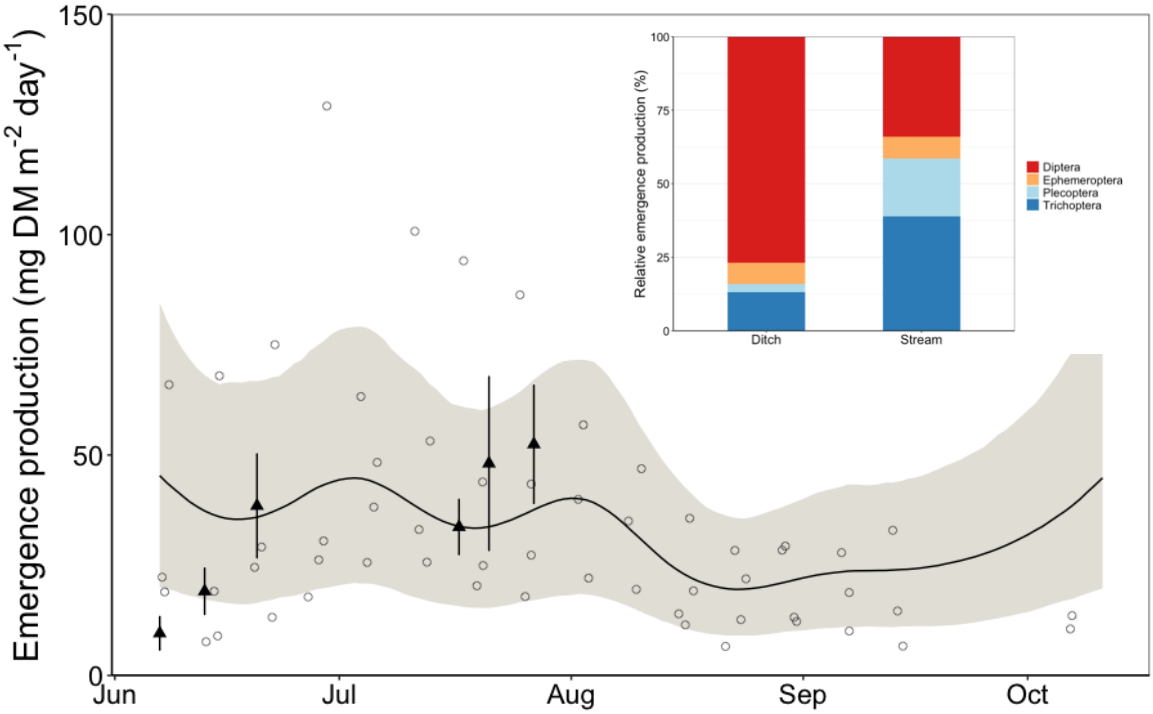

Figure 5. Aquatic insect emergence (mg DM m^{-2} d^{-1}) in irrigation ditches and streams in the Gallatin River Valley, Montana. Triangles and error bars represent the mean and standard error of emergence production in streams. Open circles represent the mean emergence from irrigation ditches, with two values that exceeded 250 mg m^{-2} d^{-1} in August not shown on this scale. The solid line represents the fitted daily emergence production in irrigation ditches using a generalized additive mixed model. The light gray shading is the 95% credible interval. The inset shows the average percent contribution of the major insect orders to emergence in irrigation ditches and streams.

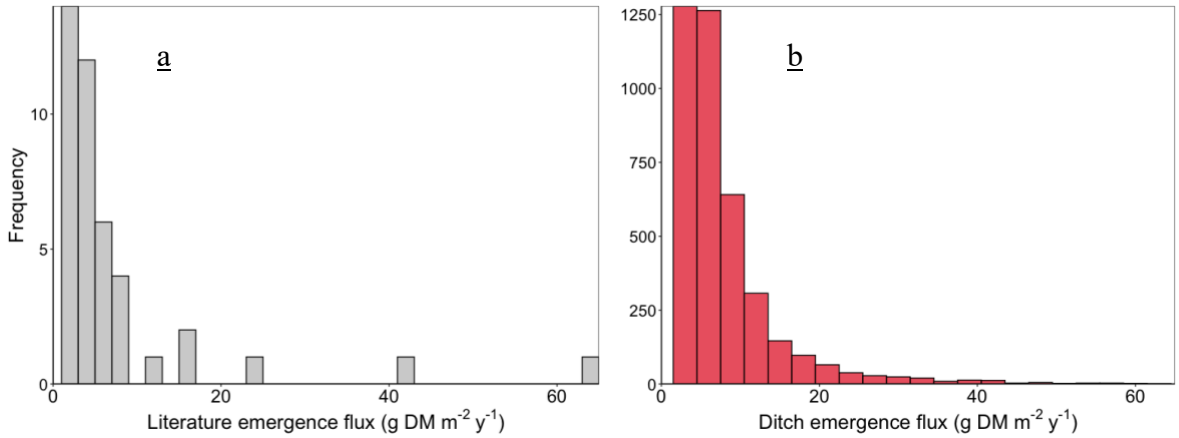

Figure 6. Frequency distributions of annual emergence flux (g DM m^{-2} y^{-1}) from (a) published values in streams (see Table S2) and (b) modeled estimates from irrigation ditches in the Gallatin River Valley, Montana using a generalized additive mixed model. Note different scales on the y-axis.

Comparing Emergence Flux Among Ditches

Study ditches differed in their magnitude and phenology of daily emergence production (Figure 7), demonstrating asynchronous emergence both in space and time. Median flux in Lowline Canal increased more than fivefold from June to July (15 to 80 mg DM m^{-2} d^{-1}), just before the cessation of water diversion in August. In contrast, emergence from Beck Border Ditch and Farmer's Canal varied less and averaged between 10 to 30 mg DM m^{-2} d^{-1}. Beck Border reached peak emergence flux (70 mg DM m^{-2} d^{-1}) at the onset of the irrigation season, whereas Farmer's Canal reached peak flux (52 mg DM m^{-2} d^{-1}) towards the end. Emergence from Lower Middle Creek Supply increased throughout June and July to its peak (48 mg DM m^{-2} d^{-1}) before receding throughout the latter half of the irrigation season.

Differences in the seasonality of emergence among ditches resulted in distinct patterns of biomass export throughout the irrigation season (Figure 8). Lowline Canal had the highest overall cumulative emergence flux (4.5 g DM m^{-2} y^{-1} [95% CrI: 1.4-16.0]), despite drying well

before the other ditches due to water shortages. Over 50% of the total export in Lowline occurred from late July to early August as a large number and biomass of Diptera emerged, even after the headgate closed and flow ceased. The accumulation of emerged biomass among other study ditches was more gradual compared to Lowline, but they still exported between 3.0 and 3.5 g DM m^{-2} y^{-1} (95% CrI: 0.9-12.4) during this time (June to October), representing irrigation season contributions comparable to values from perennial streams (Table S2).

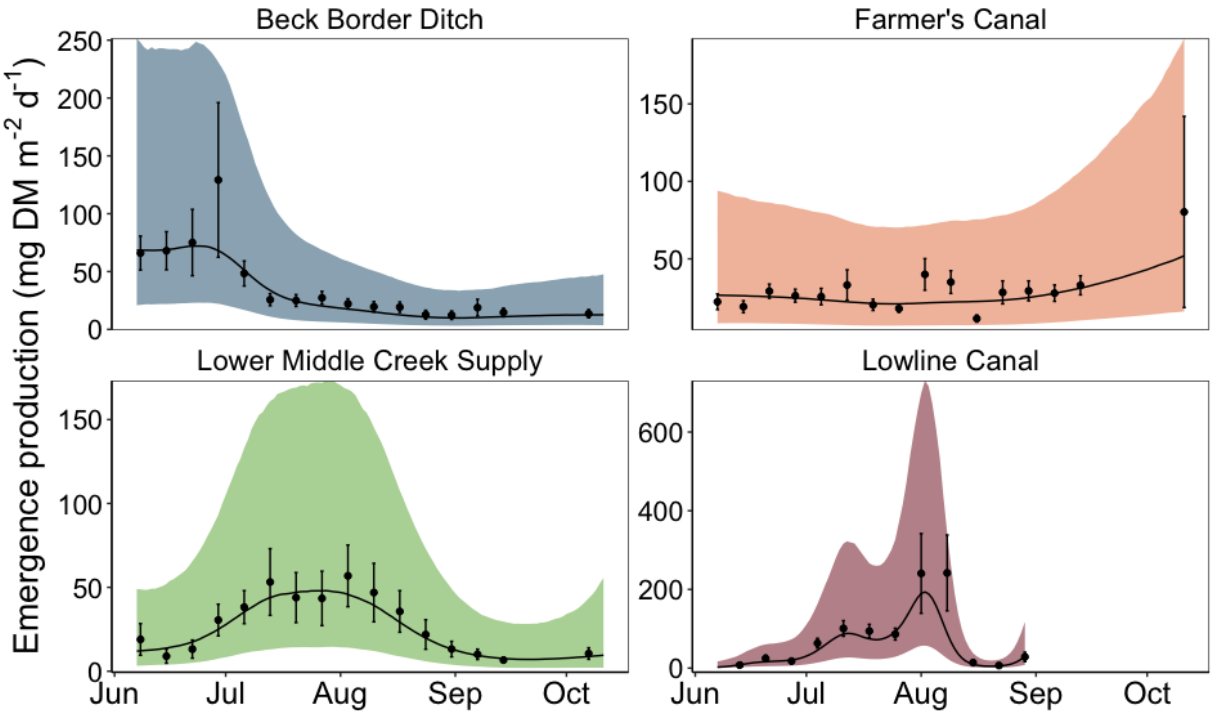

Figure 7. Fitted daily emergence production (mg DM m^{-2} d^{-1}) among irrigation ditches in the Gallatin River Valley, Montana using a generalized additive mixed model (GAMM). Solid lines represent the median emergence from the posterior distribution. Shaded areas represent the 95% credible interval. Points represent the mean and standard error of emergence production for each study ditch. Note different scales on the y-axis among panels.

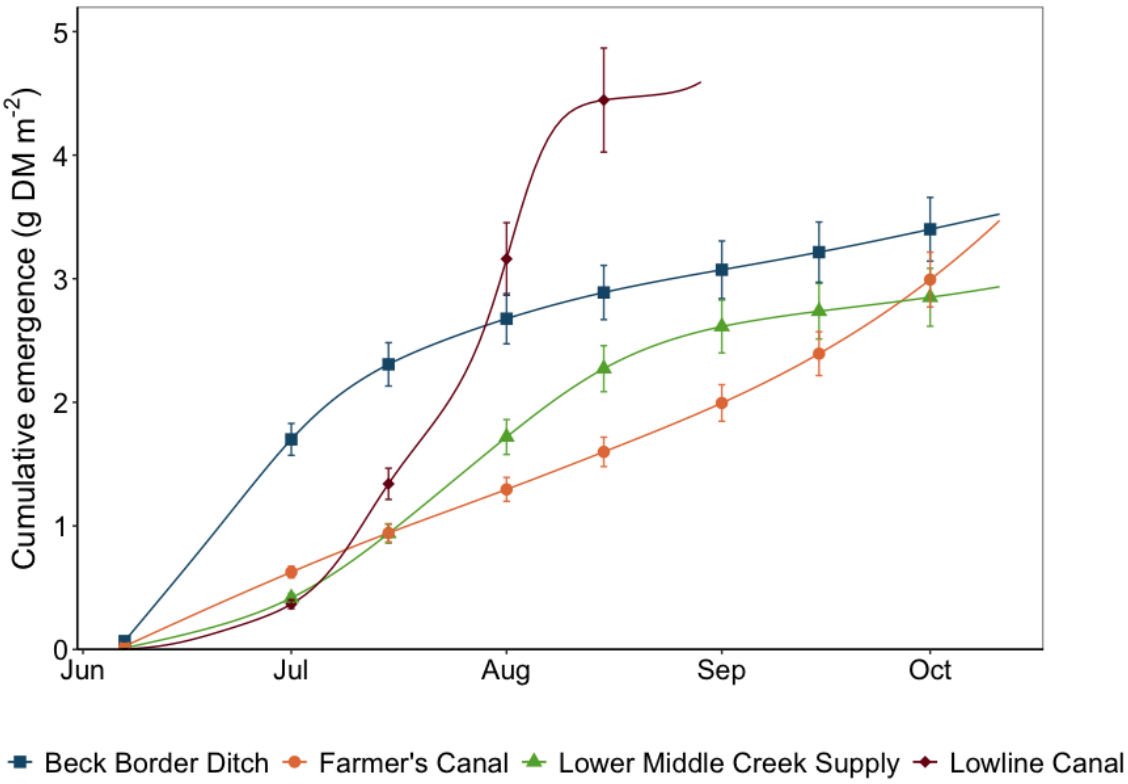

Figure 8. Emergence biomass accumulation (g DM m^{-2}) among irrigation ditches in the Gallatin River Valley, Montana. The solid line represents the cumulative sum of median flux for each day from the posterior distribution of the generalized additive mixed model. Points represent the cumulative median and standard error of emergence production on that given date. Production in Lowline Canal ceased in August as a result of water shortages among water users.

Composition of Emergence Flux

Five different orders of insects were observed emerging from focal ditches, including sensitive taxa such as the giant salmonfly, *Pteronarcys californica*. Diptera were the most abundant order and represented over 75% of the total flux from study ditches, whereas Ephemeroptera, Plecoptera, and Trichoptera (EPT) were more abundant in ditches with lower average temperature (Figure 9). Lower Middle Creek Supply was the coldest ditch (average 12.5°C) and EPT constituted 43% of the total export, whereas Lowline Canal was warmer (average 14.4°C) and EPT comprised less than 10% of the total. The emergence of EPT also

declined with distance from a natural source (beta regression, $p < 0.001$; Figure 10). At the sampling site closest to a natural source (ca. 5 m, average 11.6°C), EPT comprised 67% of total emergence production, whereas at the site farthest away (ca. 55.8 km, average 16.3°C), EPT comprised less than 1%.

The timing and magnitude of emergence varied among the different major orders and among study ditches (Figure 11), demonstrating asynchronous patterns of emergence across ditches. Diptera, as the dominant contributor to emergence flux, had a major influence on temporal patterns of total emergence. Daily Diptera emergence was substantial in Lowline Canal during the month before flow cessation (> 500 mg DM m^{-2} d^{-1} at some locations) and was consistently between 30 and 60 mg DM m^{-2} d^{-1} in other study ditches. The phenology of Plecoptera emergence was somewhat synchronous within and among ditches, peaking early in the irrigation season (late June) when water temperatures were relatively cool (10 to 15 °C) and particularly abundant at sites closer to a natural source (< 50 m away). Trichoptera flux generally varied between 10 and 30 mg DM m^{-2} d^{-1}, but was highest in Lower Middle Creek Supply in mid-July (> 100 mg DM m^{-2} d^{-1} at some locations) characterizing over 25% of the total biomass export from this ditch. Emergence flux of Ephemeroptera was low relative to other taxa (between 5 and 10 mg DM m^{-2} d^{-1}) but was widespread (occurring at most sites and ditches throughout the irrigation season), resulting in greater cumulative biomass export than Plectopera.

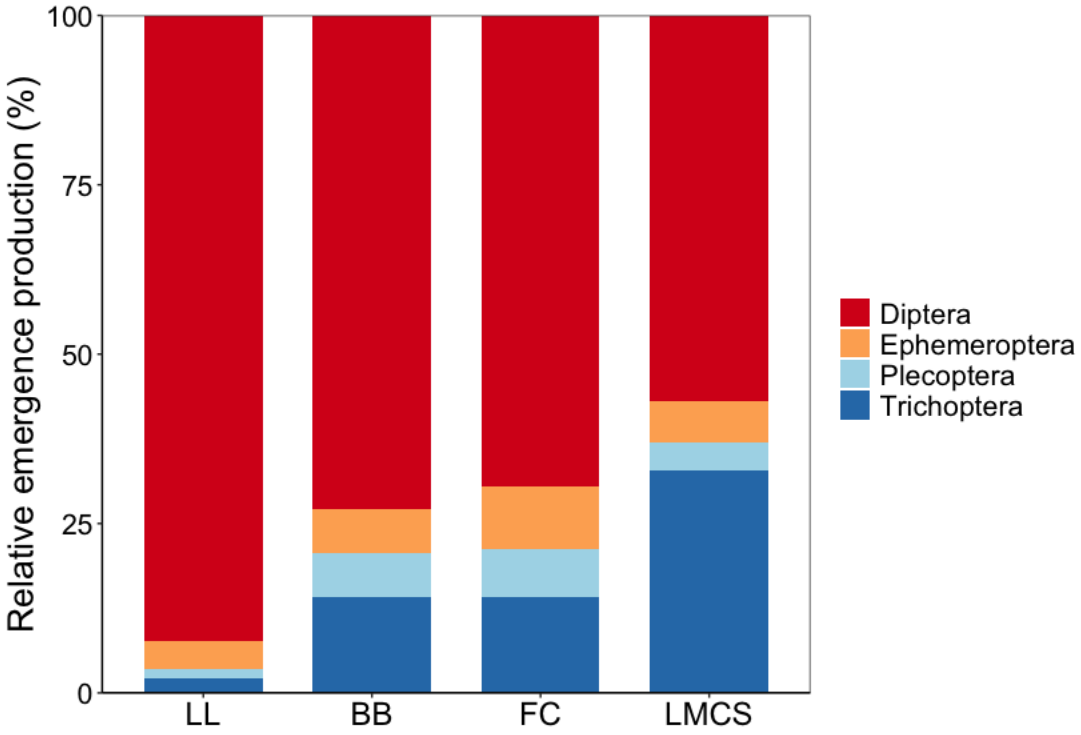

Figure 9. Percent (%) contribution of major insect orders to emergence production in study ditches. Irrigation ditch names are abbreviated, where LL is Lowline Canal, BB is Beck Border Ditch, FC is Farmer's Canal, and LMCS is Lower Middle Creek Supply.

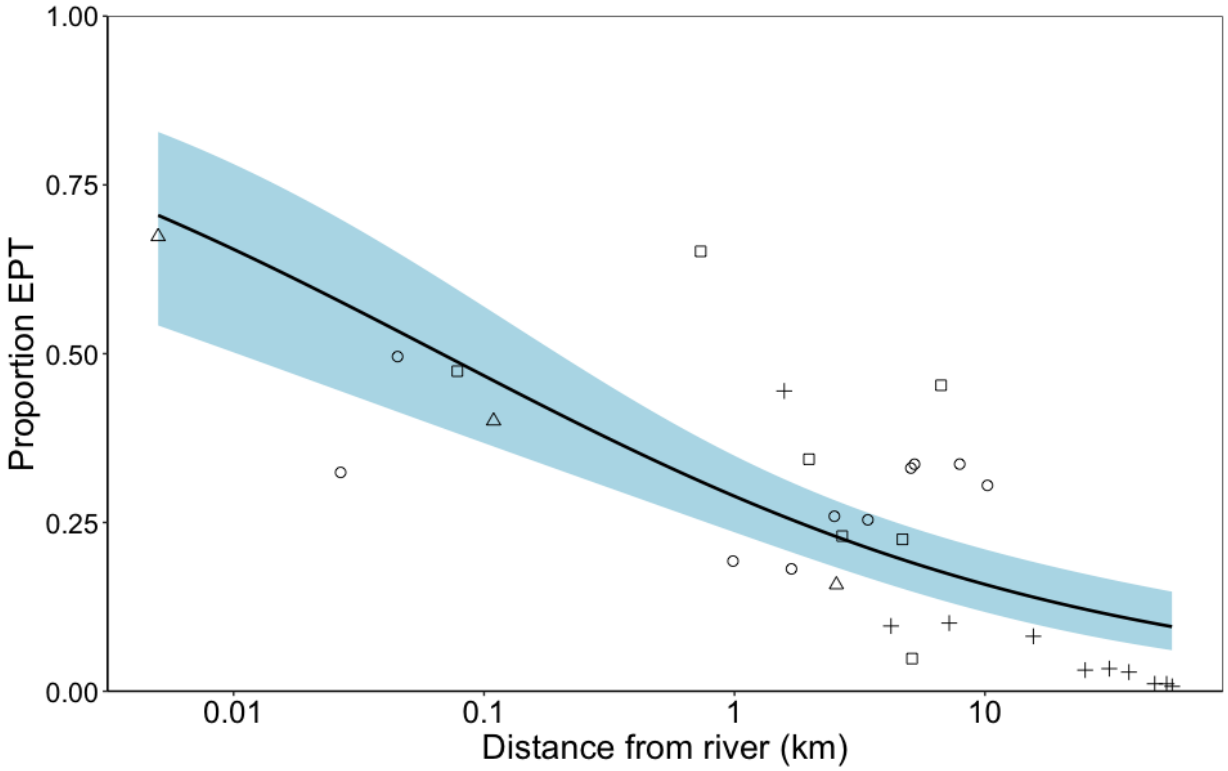

Figure 10. Proportion of ditch emergence biomass (mg DM m^{-2} d^{-1}) that is Ephemeroptera, Plecoptera, and Trichoptera (EPT) as a function of distance from a natural source (km). The black line is the fitted estimate from a beta regression model, and the blue shading is the 95% confidence interval. Shapes represent different focal ditches, where the triangle is Lower Middle Creek Supply, the box is Beck Border Ditch, the circle is Farmer's Canal, and the plus is Lowline Canal.

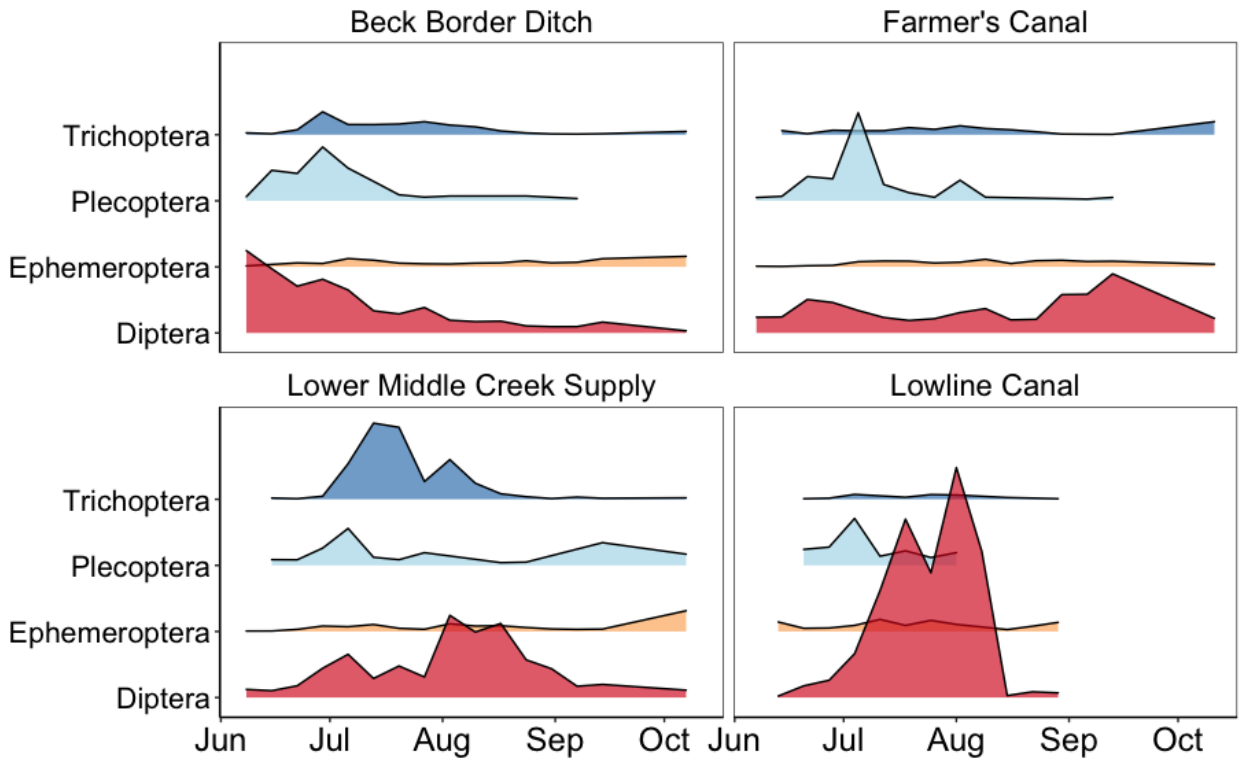

Figure 11. Emergence phenology of the dominant insect orders in study ditches. The height of the shaded area is the mean biomass flux (mg DM m^{-2} d^{-1}) on a particular date, with all heights on a common scale.

Valley-Wide Emergence Estimates

When annual emergence production (g DM m^{-2} y^{-1}) was scaled to the surface area (m^2) of lotic habitats during the irrigation season (15 May to 1 November), the ditch network was the second largest exporter of biomass in the valley (Figure 12). A median of 34 metric tons of insect biomass (95% Predictive Interval; 13-159 metric tons DM y^{-1}) was exported from the ditch network during the irrigation season. If we assume that at least 75% of insects that disperse to land do not return to the water (Gray 1989, Jackson and Fisher 1986, Petersen et al. 1999), this suggests that over 25 metric tons of insect biomass was deposited from ditches to terrestrial

habitats, 30% of the total insect deposition in the valley. The West Gallatin River constituted a majority of total biomass flux, more due to its area than to a difference in emergence biomass flux. Non-mainstem perennial streams and the East Gallatin River exported less than the ditch network, and combined, represented about a third of total export.

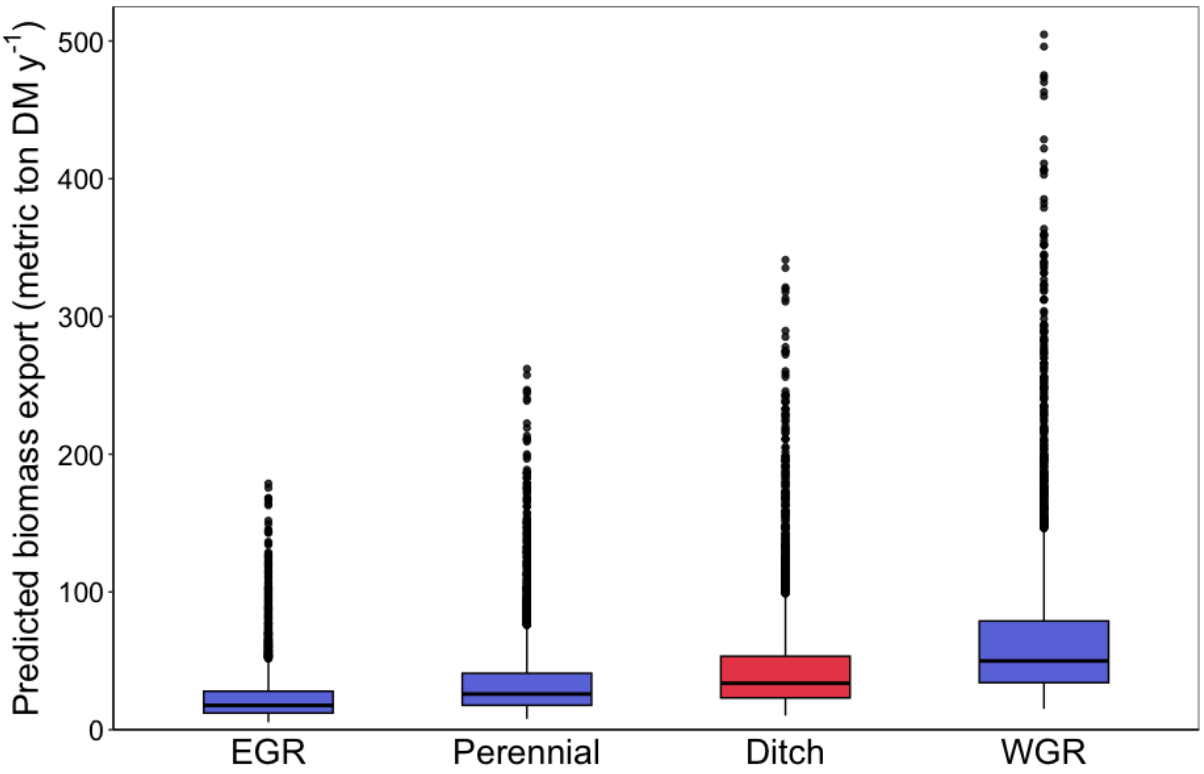

Figure 12. Distributions of predicted insect biomass export (metric tons DM y^{-1}) from lotic habitats in the Gallatin River Valley, Montana during the full irrigation season (15 May to 1 November). Box and whisker distributions represent the median (central line), interquartile range (edges of box), 2.5th and 97.5th percentiles (ends of whiskers), and values outside of this range (points). Dark blue boxes represent perennial streams and red represents irrigation ditches. We preformed this analysis separately for the two largest perennial streams in the valley, the East Gallatin River (EGR) and the West Gallatin River (WGR).

Climate Change and Water Use Scenarios

Scenarios that considered potential climate-induced changes to duration and extent of the irrigation season significantly modified patterns of insect emergence and corresponding biomass export from the ditch network (Figure 13). Under changes in ditch use due to the driest conditions, insect emergence is predicted to decline by over 70% relative to measured values in 2022. This large reduction resulted from additive effects of a shorter irrigation season and a large reduction in the areal wetted extent of the ditch network. Expected changes in ditch use during a typical dry year led to a 36% reduction in emergence, only affecting emergence from ditches used by irrigators with junior water rights. In contrast, under a scenario where ditch use is extended and unimpeded by stream flow, biomass export was predicted to increase by 18%.

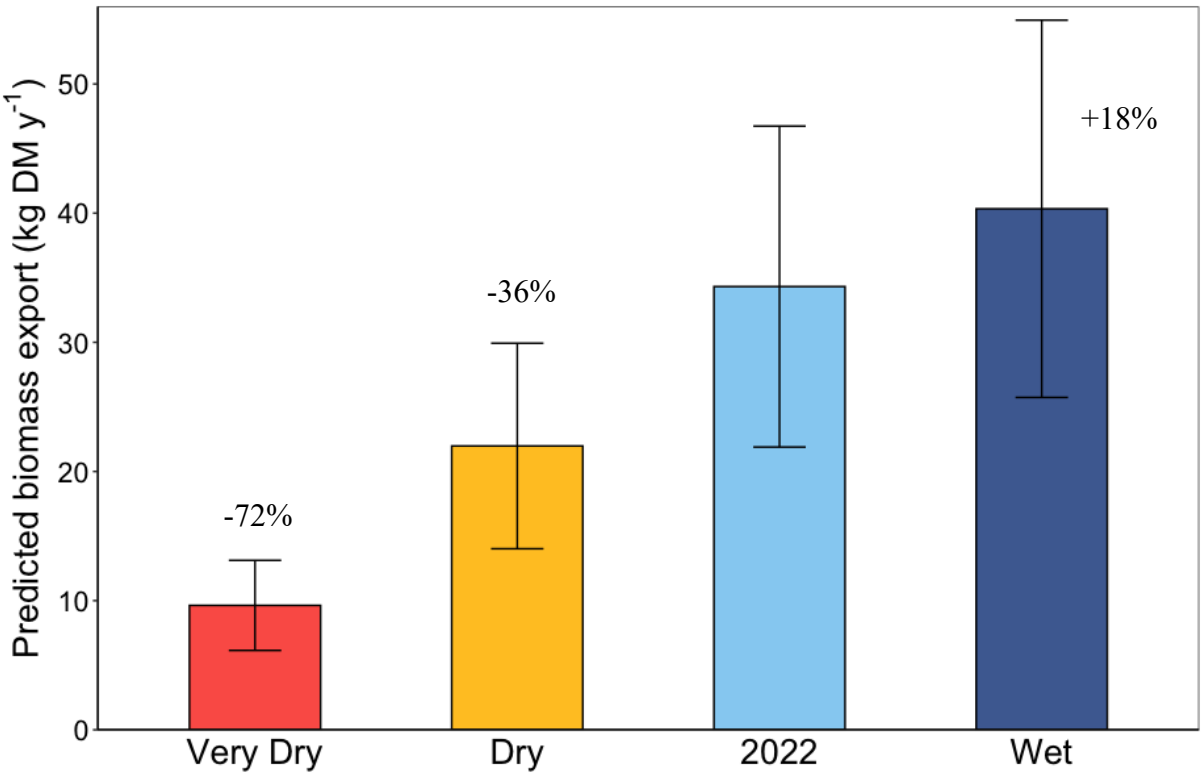

Figure 13. Predicted insect biomass export (metric tons DM y^{-1}) from the ditch network during the 2022 irrigation season (171 days of ditch use) and under three water availability scenarios: very dry (48 days of ditch use), dry (48 days of ditch use for ditches with junior rights and 171 days of ditch use for ditches with senior rights), and wet (201 days of ditch use). Error bars represent simulated variation in ditch network surface area. Numbers above bars represent the predicted percent change in biomass export relative to our study year.

DISCUSSION

Irrigation ditches represent ubiquitous but underappreciated non-perennial stream ecosystems that extend the reach and magnitude of energetic subsidies in intermountain basins where irrigation withdrawals occur. Our study demonstrates that these ditches provide artificial habitat that supports diverse aquatic insect communities and, through adult insect emergence, contribute large energetic subsidies to terrestrial ecosystems. We found that the median annual emergence flux from ditches in the Gallatin Valley was substantial (~6 g DM m^{-2} y^{-1}), and actually exceeded that of non-mainstem streams in the valley network during the extent of the irrigation season. We also found that heterogeneity in flow permanence and temperature among sites created a mosaic of environmental conditions that led to large variation in the phenology, composition, and magnitude of aquatic insect emergence, contributing to a more stable portfolio of energetic subsidies across the surface-water network. Given the growing demands for water in arid regions, and changes in snowpack and flow regimes driven by climate change and water withdrawals (Dettinger et al. 2015, Siirila-Woodburn et al. 2021), it is likely that ecosystem processes and services driven by agricultural ditches will change over time. It is thus important that we understand their influence on, and contribution to, regional biodiversity and ecological processes in intermountain basins of the New West.

Our comparisons of ditches and streams are in line with previous studies that demonstrate agricultural streams can have just as much production of adult insects (if not more) than less impaired streams (Krell et al. 2015, Raitif et al. 2018, Ohler et al. 2023). Although agricultural land use typically reduces the average body size of benthic insects by shifting communities towards smaller and more tolerant taxa (e.g., Chironomidae, Stenroth et al. 2014), the increase in

sheer number of emerging adults can drive greater overall magnitude of production and emergence subsidies. Thus, from a biomass perspective, irrigation ditches may have a disproportionate effect on ecological processes in the Gallatin River Valley, possibly contributing to resources for terrestrial insectivores or providing other critical ecological functions in agroecosystems (e.g., pollination, soil fertilization, and pest control; see Raitif et al. 2019). At an annual scale, emergence production in ditches was similar to values published in perennial streams, demonstrating elevated daily production values that remained steady throughout the irrigation season. Therefore, emergence from ditches compliments subsidies provided by nearby natural streams at a level similar to, or potentially greater, than periods before water withdrawal in the valley.

Drying is typically considered a disturbance in stream ecosystems (Lake 2003), and the degree of hydrologic connectivity to perennial streams may thus be an important driver of variation in community structure and processes in intermittent streams (Boultan 2003, Bogan et al. 2013, Leigh and Datry 2016, Sarremejane et al. 2021, Fournier et al. 2022). In our study, we predicted that the number of insect orders represented in emergence would decline with distance from a perennial source, with the caveat that ditches have been in place long enough to favor taxa that are resistant to drying (Fiege 1999, Arumí et al. 2009). We found that the composition of emergence was more diverse and contained larger-bodied taxa that are less tolerant to lower oxygen conditions (i.e., EPT) at sites closer to perennial streams. These sites also contained benthic substrates more typical of natural streams in the valley (i.e., gravel), and temperatures that were similar to natural streams, or even cooler. In contrast, sites farther away from perennial sources supported small-bodied Diptera, but their frequent and abundant emergence resulted in

higher estimates of biomass flux. These sites also tended to have warmer temperatures and more homogenous substrates (largely sand), reflecting a more managed artificial system put in place solely for irrigation. Thus, connectivity to perennial sources and degree of artificiality created heterogeneity in emergence character and magnitude across the ditch network, which likely influenced the pathways and fate of this subsidy in terrestrial environments.

We found the study ditch that dried during the irrigation season (i.e., Lowline Canal) to have the greatest daily emergence production and cumulative biomass export. This result was surprising given that many previous studies have suggested that invertebrate abundance and richness should decline with intensity of drying (Datry et al. 2017). In our case, other unmeasured factors may have led to increased emergence production, including a lack of aquatic predators or elevated temperatures and nutrients (Pope et al. 2009, Wesner 2010, Greig et al. 2011). It may also be the case that previous studies have not quantified emergence on a cumulative basis, and factors such as warm temperatures and dominance by Diptera may enhance biomass turnover leading to high levels of production. For example, Leathers et al. (2024) found that earlier low flows resulted in warmer temperatures and a near doubling of dipteran emergence. Moreover, many insects concentrate in pools during flow cessation and some disperse as adults to avoid desiccation (Gray and Fisher 1981, Bunn and Hughes 1997, Bogan and Boersma 2012). Such a response to drying may create large, synchronized pulses of biomass (i.e., temporal hotspots) that influence foraging and fitness of terrestrial consumers that concentrate near these pools.

We found clear differences in emergence phenology and duration among ditches, likely resulting from differences in environmental conditions and community composition. While

emergence from Farmer's Canal was relatively consistent throughout the irrigation season, Lowline Canal showed a distinct peak in late July, followed by an abrupt decrease into September. These observed differences were likely attributed to variation in water temperature, substrates, and degree of mixing with perennial sources. Farmer's Canal maintained cooler temperatures and supported consistent emergence of the major insect orders throughout the irrigation season. In contrast, Lowline Canal, with temperatures ~2-4°C warmer and an earlier dewatering, experienced little emergence of classically sensitive taxa (EPT) but significant Diptera emergence, especially in hot summer months prior to the closing of the headgate. Previous studies have postulated that human-modification and climate change may synchronize emergence across space and time and limit the duration of resource availability for recipient consumers (Schindler and Smits 2017, Nash et al. 2023). However, we observed that thermal, substrate, and flow heterogeneity across the ditch network created a robust "portfolio" of aquatic subsidies at the landscape-level, providing stable resources for terrestrial consumers that can move across the landscape (Uno 2016, Schindler and Smits 2017)

Many studies have quantified variation in local emergence production fluxes, but few have attempted to scale these measurements to regional systems (Gratton and Vander Zanden 2009, Bartrons et al. 2013, Dreyer et al. 2015, Wesner et al. 2019). To understand regional contribution and relative uncertainty of stream network components (e.g., intermittent or perennial, natural or artificial) must incorporate variability in emergence and wetted surface area through time. We incorporated variation in wetted width throughout the irrigation season in ditches, and uncertainty in emergence production through a comprehensive sampling of 30 sites that varied strongly in environmental conditions. The result was large relative uncertainty in our predicted

annual emergence values (i.e., 2 to 27 g DM m^{-2} y^{-1}; Figure 6) that propagated to estimates at the network scale (13 to 159 tons DM y^{-1}; Figure 12). Nonetheless, we suggest that attempts to constrain the magnitude of aquatic subsidies with attention to propagation of uncertainty is a critical first step to understanding the role of different components (such as perennial or intermittent networks) to regional ecosystem processes.

Although quantification of aquatic subsidies at large scales introduces substantial uncertainties, Wesner et al. (2019) used a similar emergence model and bioenergetic information to estimate the number of birds that could be supported by aquatic insects emerging from Missouri River backwaters. Using his approach, and assuming an energetic requirement of 3800 kJ d^{-1} ha^{-1} and mean density of 59 birds ha^{-1}, the Gallatin Valley ditch network could theoretically support the energetic demands of 12,000 to 330,000 birds (median of 70,000) throughout the irrigation season (May to November). In our study, we did observe riparian birds (most notably yellow warblers and tree swallows) and orb-weaving spiders consuming emerging insects from ditches, as well as other birds directly foraging in ditches (dipper and killdeer). These observations and model estimates reinforce the idea that artificial ditches have a high potential to influence broader community- and ecosystem-level processes.

Water is already a scarce resource in agricultural regions of the western US, and projected shifts in snow-pack magnitude and melt timing will only exacerbate demand (Dettinger et al. 2015, Siirila-Woodburn et al. 2021). Similar to ecosystem processes in natural intermittent streams (Sarremejane et al. 2024), declines in hydroperiod across ditch networks to climate change are likely to reduce the magnitude, duration, and subsequent consequence of aquatic subsidies. Recent interest in ‘designer’ flows in flow-modified river networks (Chen and Olden

2017) could benefit from considering the role irrigation ditches in providing artificial habitat and subsequent energetic subsidies to terrestrial ecosystems. To date, water abstraction for agricultural purposes has been thought as a major cause of anthropogenic flow intermittence (AFI), in which human activities and consequent changes in hydrology lead to dewatering and intermittence of previously perennial systems (*sensu* Datry et al. 2023). While AFI may significantly reduce the total size of river networks (McIntosh et al. 2024) and potentially shift biotic communities towards new ecosystem states (Larned et al. 2010, Zipper et al. 2020, 2022), in many western US river valleys with snowmelt-dominated hydrographs, water withdrawals from mainstem rivers inundate vast numbers of irrigation ditches. This provides 'engineered' surface water in the floodplain at a time (i.e., spring-early summer) that corresponds with snowmelt events and during periods of low flow in the summer when water is historically less abundant (*but see* Naiman et al. 1988). Because humans, and the water rights that govern them, control the timing and distribution of surface water within irrigation ditches, they create an artificial network that is hydrologically distinct from natural intermittent networks (Carlson et al. 2019). The resulting novel extent of lotic ecosystem could influence regional biodiversity and ecosystem function (Acuna et al. 2014, Datry et al. 2017), or just provide new habitat otherwise not available. As water scarcity and anthropogenic drying become more prevalent across historically perennial river networks, irrigation ditches provide a natural experiment for understanding how biota may respond to varying degrees of artificial drying (Koetsier and McCauley 2015).

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APPENDIX

SUPPLEMENTARY INFORMATION

Justification of Priors in Bayesian Models

Because the predictor (days) is centered in the GAMM, the intercept in our model represented mean daily insect emergence (mg DM m^{-2} d^{-1}) near the start of June. To determine this prior, we calculated the average daily emergence from 32 publications of annual emergence by dividing annual emergence by 365 days (Table S2). The result was mean daily emergence of 8.0 mg DM m^{-2} d^{-1} with a standard deviation of 11.3. Therefore, we chose normal(log(10),log(50)) as a reasonable estimate that was somewhat lower than expected peak emergence, but included anywhere from zero emergence (or below) to >100 mg DM m^{-2} d^{-1}. Because the expected outcome in the model is filtered through a log-link, it can take on negative and continuous values. Therefore, a normal prior distribution seemed like a reasonable choice. Other prior specifications were chosen as vague priors following the approach in Wesner et al. (2019).

Estimating Emergence at Times Before and After Our Sample Dates

To approximate the amount of emergence we missed by not sampling before 7 June or after 11 October, we used median emergence on our first and last sample date (45.4 and 45.7 mg DM m^{-2} d^{-1}, respectively), multiplied by 44 to simulate 44 days of additional emergence on either end of our sampling period. We multiplied by 44 days because we assumed the irrigation season in the Gallatin Valley is 15 May to 1 November, and we missed the first 23 days between 15 May and 7 June and the last 21 days between 11 October and 1 November. We decided to use the median emergence on the first and last date to extrapolate because it occurred in the middle of increased trends in each direction (i.e. increasing towards earlier dates, and increasing towards

later dates, Figure 5), and we therefore thought this would be a conservative approach. Alternatively, we could have extended the predictions from the model which would have yielded higher annual production.

Using the methods above, we estimated that an additional 44 days of sampling before and after our collection periods would have yielded a median of 2.0 g DM m^{-2} y^{-1} [0.9-5.0, 95% CrI]. We therefore added this amount to the estimate from Table S4 to quantify the annual production of the ditch network.

Table S1. Physical characteristic and water right of the focal irrigation ditches within the Gallatin Valley ditch network, Montana. Descriptions of each factor are below the table.

	Beck Border Ditch	Farmer's Canal	Lower Middle Creek Supply	Lowline Canal
Total length[1]	14.1	16.5	7.8	60.0
Number of intersections[2]	4	3	2	0
Number of Sites[3]	7	10	3	10
Site range[4]	0.7 – 6.7	0.2 – 10.3	0.005 – 2.5	1.5 – 55.8
First decree year[5]	1866	1866	1865	1890
Company establishment[6]	1891	1889	1890	1924
Number of drying events[7]	0	0	0	2

[1]Total length (km) is the length of the ditch from where it diverts water from the Gallatin River to where it ends.
[2]Number of intersections is the number of perennial streams the ditch intersects along the course of its flow.
[3]Number of sites is the amount of aquatic invertebrate sites within each ditch.
[4]Site range (km) is the minimum and maximum distance a site was from a natural river.
[5]The first decree year is the date of the first water right within each ditch.
[6]Company establishment is the year the ditch company (i.e., entity operating the ditch) was established.
[7]Number of drying events is how many times the ditch dried during the irrigation season (15 May to 1 November).

Table S2. Mean annual (g DM m^{-2} y^{-1}) and daily emergence production (mg DM m^{-2} d^{-1}) of emerging aquatic insects from published values in streams.

Reference	Location	Annual emergence (g DM m^{-2} y^{-1})	Daily emergence (mg DM m^{-2} d^{-1})
Illies 1971	Breitenback Stream, Germany	3.7	10.2
Ringe 1974	Rohrweisenback Stream, Germany	3.2	8.7
Ringe 1974	Breitenback Stream, Germany	5.4	14.9
Bottger 1975	Kalengo Stream, Zambia	4.0	10.8
Gumbel 1976	Kalkbach Stream, Germany	2.0	5.6
Gumbel 1976	Kalkbach Stream, Germany	2.6	7.1
Gumbel 1976	Breitenback Stream, Germany	2.6	7.2
Gumbel 1976	Breitenback Stream, Germany	3.7	10.2
Harper 1978	I'Achigan River, Quebec	5.3	14.5
Harper 1978	I'Achigan River, Quebec	7.1	19.4
Jackson and Fisher 1986	Sycamore Creek, Arizona	23.1	63.3
Gray 1989	Kings Creek, Kansas	7.4	20.3
Malison et al. 2010	Darby River, Idaho	5.1	14.0
Malison et al. 2010	Fox River, Idaho	2.2	6.0
Malison et al. 2010	NF Mahogany River, Idaho	3.2	8.7
Malison et al. 2010	South Leigh River, Idaho	7.0	19.0
Malison et al. 2010	SF Mahogany River, Idaho	4.1	11.1
Malison et al. 2010	Teton River, Idaho	8.1	22.2
Small et al. 2013	Arboleda Stream, Costa Rica	1.4	3.7
Small et al. 2013	Sura Stream, Costa Rica	1.6	4.3
Small et al. 2013	Saltito Stream, Costa Rica	3.3	9.0
Small et al. 2013	Carapa Stream, Costa Rica	3.8	10.3
Small et al. 2013	Piper Stream, Costa Rica	1.4	3.7
Hagen and Sabo 2014	Sycamore Creek, Arizona	63.1	172.9
Hagen and Sabo 2014	San Pedro River, Arizona	9.0	24.7
Heinrich et al. 2014	Upper Cache River, Illinois	16.8	46.0
Heinrich et al. 2014	Upper Cache River, Illinois	15.5	42.5
Yuen and Dudgeon 2016	Lead Mine Pass Stream, China	1.2	3.2
Yuen and Dudgeon 2016	Tai Po Kau Forest Stream, China	2.0	2.0
Raitif et al. 2018	Multiple watersheds, France	3.8	10.4
Bonjour et al. 2020	Kings Creek, Kansas	3.3	8.9
Naslund et al. 2021	Mud River, West Virginia	11.5	31.5

Table S3. Mean and standard error of the major orders of aquatic insect emergence flux (mg DM m^{-2} d^{-1}) and abundance (ind. m^{-2} d^{-1}) between irrigation ditches and streams in the Gallatin River Valley, Montana from June to August.

	Ditch flux (mg m^{-2} d^{-1})	**Stream flux (mg m^{-2} d^{-1})**	**Ditch density (ind. m^{-2} d^{-1})**	**Stream density (ind. m^{-2} d^{-1})**
Diptera	74.4 ± 7.0	24.5 ± 3.5	207 ± 19.4	98.2 ± 15.1
Trichoptera	27.9 ± 4.4	38.9 ± 9.2	19.2 ± 4.0	36.4 ± 10.7
Ephemeroptera	9.2 ± 0.7	6.8 ± 2.2	6.2 ± 0.5	7.6 ± 2.1
Plecoptera	32.3 ± 3.9	50.5 ± 13.1	3.3 ± 0.4	4.4 ± 1.2

Table S4. Annual production (g DM m^{-2} y^{-1}) estimates when leaving out one of the four ditches from the generalized additive mixed model (GAMM). Summary statistics are from the posterior predictive distribution (median and 95% predictive interval). Output is from a condensed irrigation season (7 June to 11 October).

		Annual production (g DM m^{-2} y^{-1})		
Model	Sites	low95	median	high95
GAMM	all sites	0.5	4.3	25.4
GAMM	no_LL	0.7	3.4	16.2
GAMM	no_FC	0.5	4.0	25.6
GAMM	no_BB	0.3	4.2	28.2
GAMM	no_LMCS	0.3	4.3	31.0

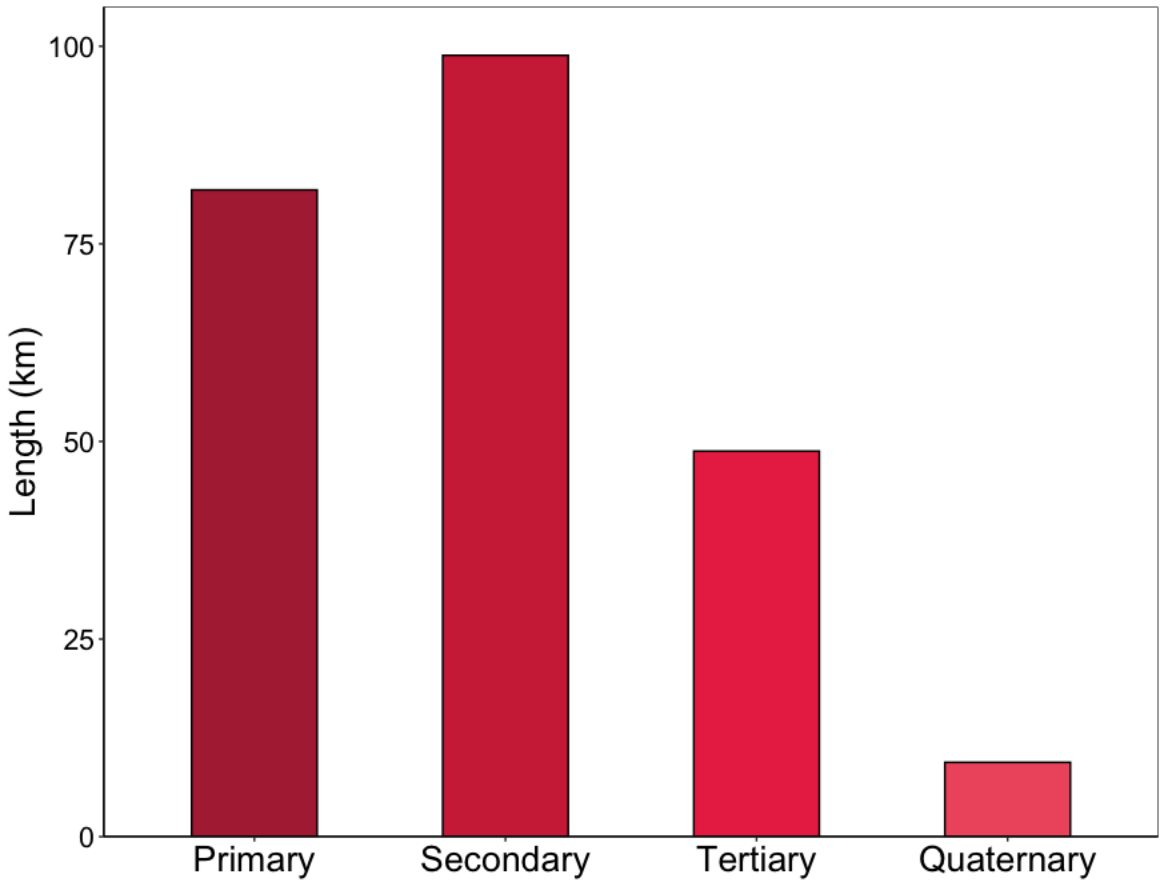

Figure S1 The length (km) of distributary branches of a major ditch in the Gallatin River Valley, Montana (see Figure 1 for illustration).

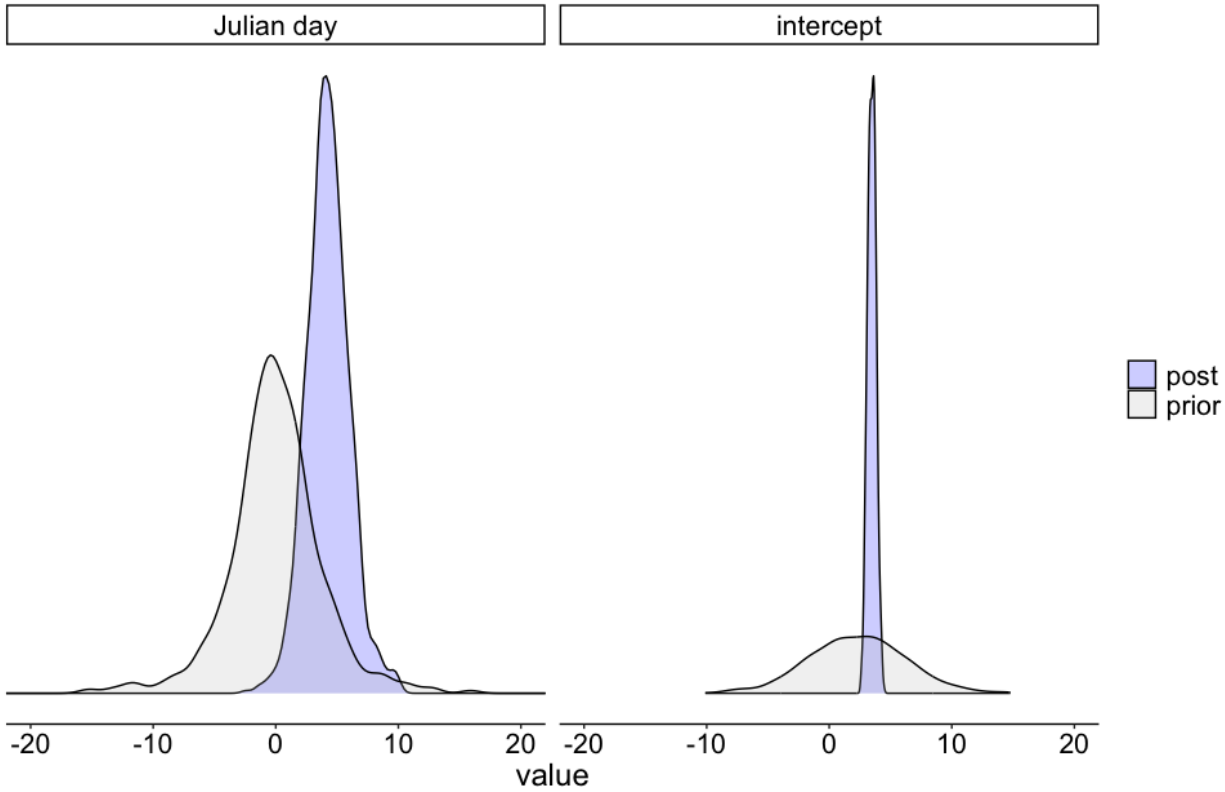

Figure S2. Comparison of prior and posterior distributions for the smoothing term (Julian day) and the intercept from the Bayesian generalized additive mixed model used to predict daily emergence flux in irrigation ditches.

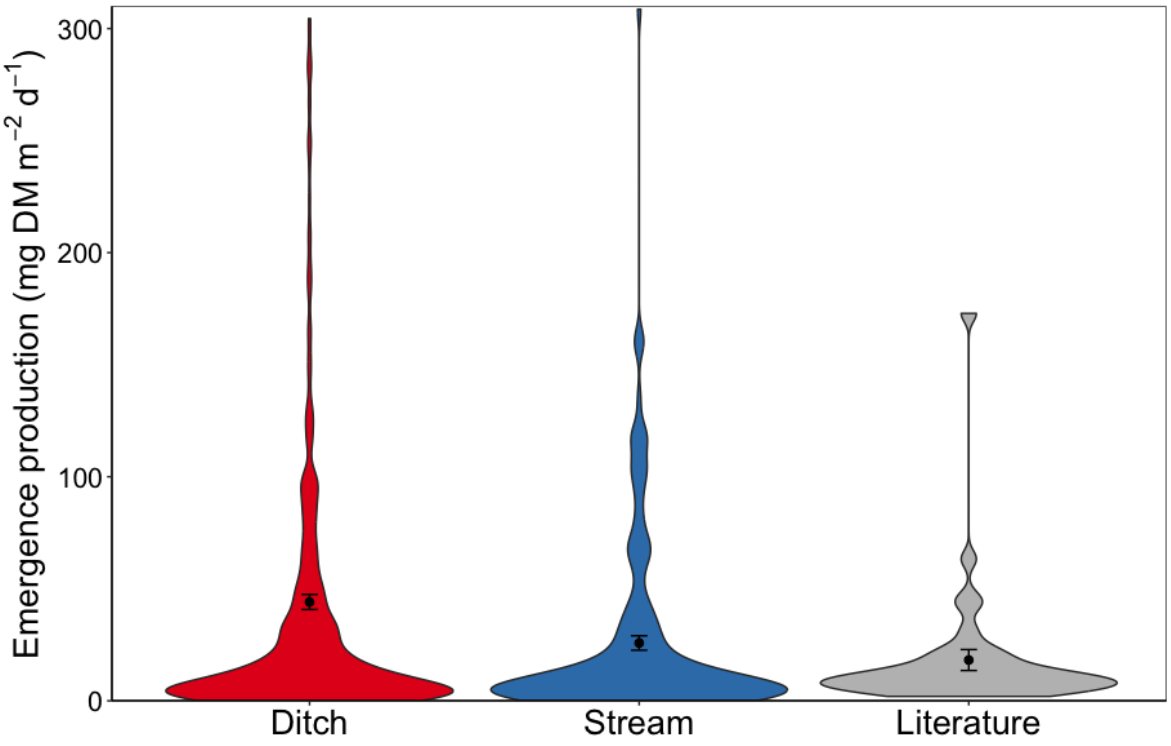

Figure S3. Violin plot of daily emergence flux (mg DM m^{-2} d^{-1}) between irrigation ditches and streams in the Gallatin River Valley, Montana, and estimates from the literature (see Table S2). Note that some production values in ditches and streams exceed y-axis limit.

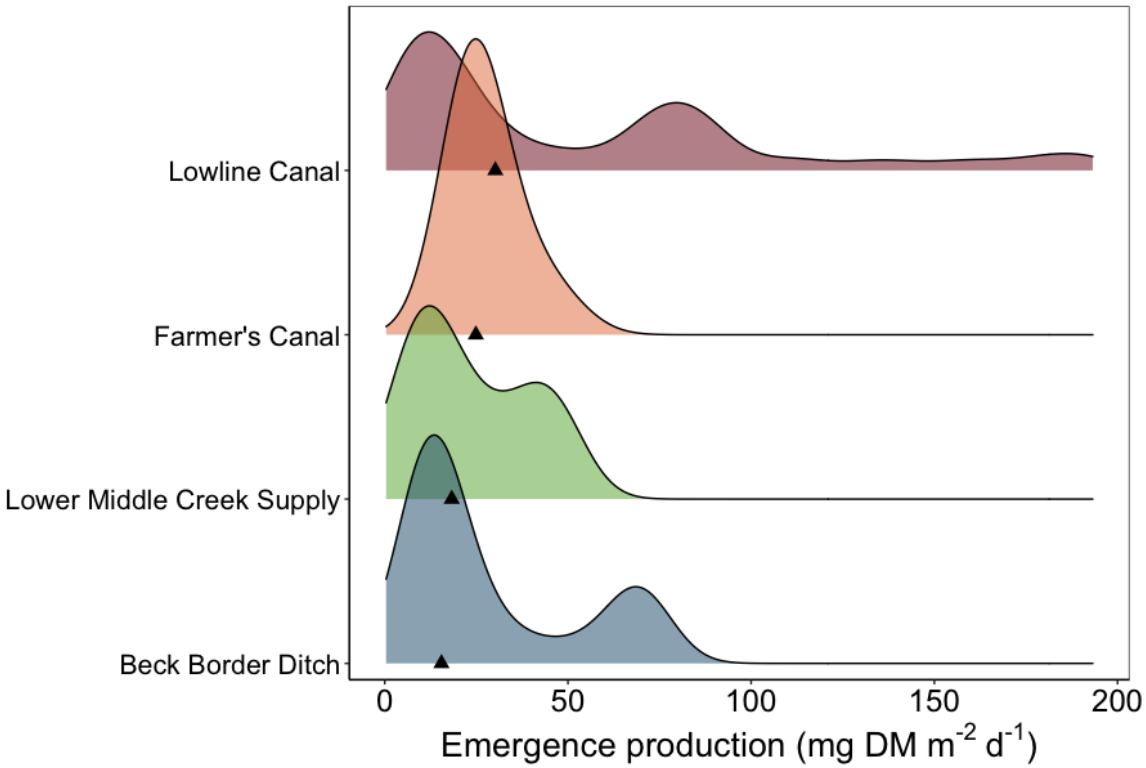

Figure S4. Posterior distributions of emergence flux (mg DM m^{-2} d^{-1}) among irrigation ditches in the Gallatin River Valley, Montana using a generalized additive mixed model. The triangle represents the median production.

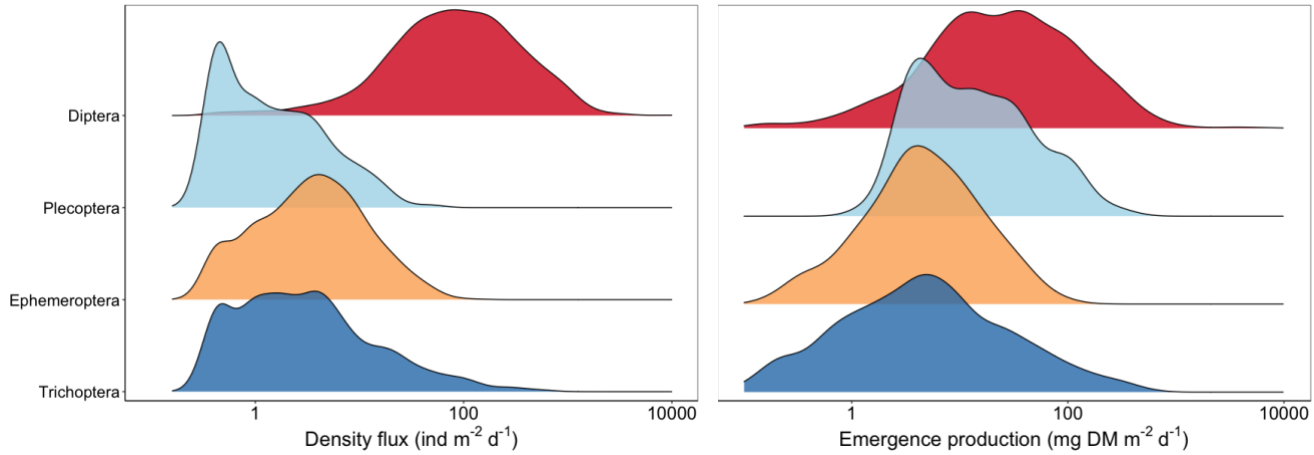

Figure S5. Distributions of emergence density (ind. m^{-2} d^{-1}) and production (mg DM m^{-2} d^{-1}) of the major insect orders from study irrigation ditches in the Gallatin River Valley, Montana.

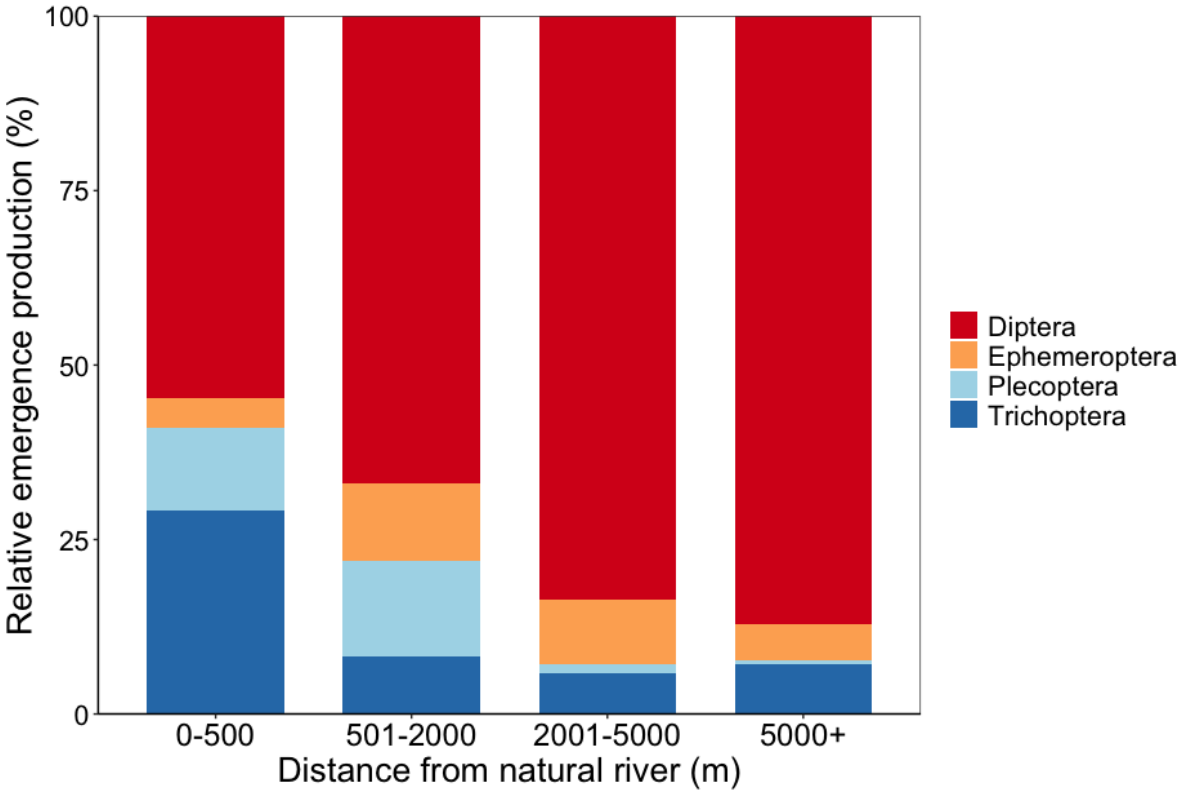

Figure S6. Percent (%) contribution of major insect orders to emergence production as a function of distance from river (m).