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Agricultural land use and macroinvertebrate assemblages in lowland temporary streams of the Willamette Valley, Oregon, USA

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ABSTRACT

Streams that dry during part of the year are common throughout the world, yet studies of the macroinvertebrate assemblages in these types of streams are rare compared to those in permanent streams; and studies that assess the effects of agriculture on temporary stream invertebrates are even rarer. We studied macroinvertebrate assemblages in lowland temporary streams of a region with high agricultural land use, the southern Willamette Valley, Oregon, USA. Overall assemblages were dominated by non-insects, and invertebrates tolerant of organic pollution. Nonetheless, these invertebrates displayed adaptations to life in temporary habitats, and as such they may be unique to temporary streams and seasonal wetlands, providing an important addition to regional biodiversity. Stream invertebrates are also important as a prey base for native fish and amphibians using these channels. Benthic invertebrate densities were higher at sites with slower water and more in-stream vegetation; to a lesser degree greater agricultural land use was associated with lower densities. Taxon richness was also negatively affected by agriculture, but this was most evident when least disturbed and highly agricultural sites were compared. Sites in watersheds with a lower proportion of their area under agriculture (mostly west of the Willamette River) had a variety of taxa in disturbance-sensitive insect orders Ephemeroptera, Plecoptera, and Trichoptera (EPT), plus flies in the family Simuliidae present. In addition, they had greater relative abundances of 2 types of flies in the family Chironomidae. In contrast, sites in watersheds with high agricultural land use (mainly east of the Willamette River) had greater relative abundances of non-insects, including ostracods, nematodes, and oligochaete worms. In highly agricultural watersheds, when stream-bottom vegetation was abundant, it was associated with greater benthic invertebrate density, but not with higher taxon richness. Our results suggest that increasing stream-bottom vegetation could be useful when food is limiting for native vertebrates. On the other hand, reduced agricultural land use allows for the development of more diverse benthic invertebrate assemblages.

1. Introduction

Streams that dry during part of a year are common throughout the world and naturally occur where rainfall is limited or highly seasonal, or where porous geology allows for the loss of surface flow (Dieterich and Anderson, 1995; Paltridge et al., 1997; Uys and O'Keefe, 1997; Meyer and Meyer, 2000; Aguiar et al., 2002; Datry et al., 2014; Mazor et al., 2014). These types of streams can vary in the length and predictability of their dry phase. Some, but not all, have permanently flowing headwaters, and they may or may not be used by fish during the wet phase. The terms ephemeral, temporary, and intermittent have been applied variously to streams that do not flow continuously. Our study focused on temporary streams, defined by Dieterich and Anderson (2000) as streams without permanent headwaters, but with flow persisting for >4 consecutive months per year. Based on the changing conditions in these kinds of systems, the annual cycle can be divided into 3 hydrologic stages: a running water stage with flowing water, a pool phase with standing water, and a terrestrial stage with surface water absent (Williams and Hynes, 1976; Williams, 1996). Our studies were conducted towards the end of the running water stage during late winter/early spring in temporary streams draining grass seed producing lands of the Willamette Valley, Oregon, U.S.A.

Previous studies of temporary streams in the Pacific Northwest region of North America have examined fishless sites within small drainage areas in relatively steep, forested terrain (Tew, 1971; Dieterich and Anderson, 2000; Price et al., 2003; Banks et al., 2007;

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Stewart and Anderson, 2008). In contrast, our study examined lowland temporary streams in a region with high agricultural land use. These streams supported native fish and amphibians during the wet season, serving as nursery grounds for some species (Colvin, 2006; Colvin et al., 2009). As invertebrates are essential prey for these and other vertebrate predators, invertebrate abundance or biomass are considered an important component of habitat suitability for vertebrates (Wipfli and Gregovich, 2002; Taft and Haig, 2005) in these temporary habitats.

Although temporary streams are less well studied than permanent streams, several patterns have been documented. These systems often have fewer benthic invertebrate taxa during their running phase than permanent streams (Williams, 1996; Williams et al., 2003; Storey and Quinn, 2008; Sanches-Montoya et al., 2010), and taxon richness decreases as the duration of the dry

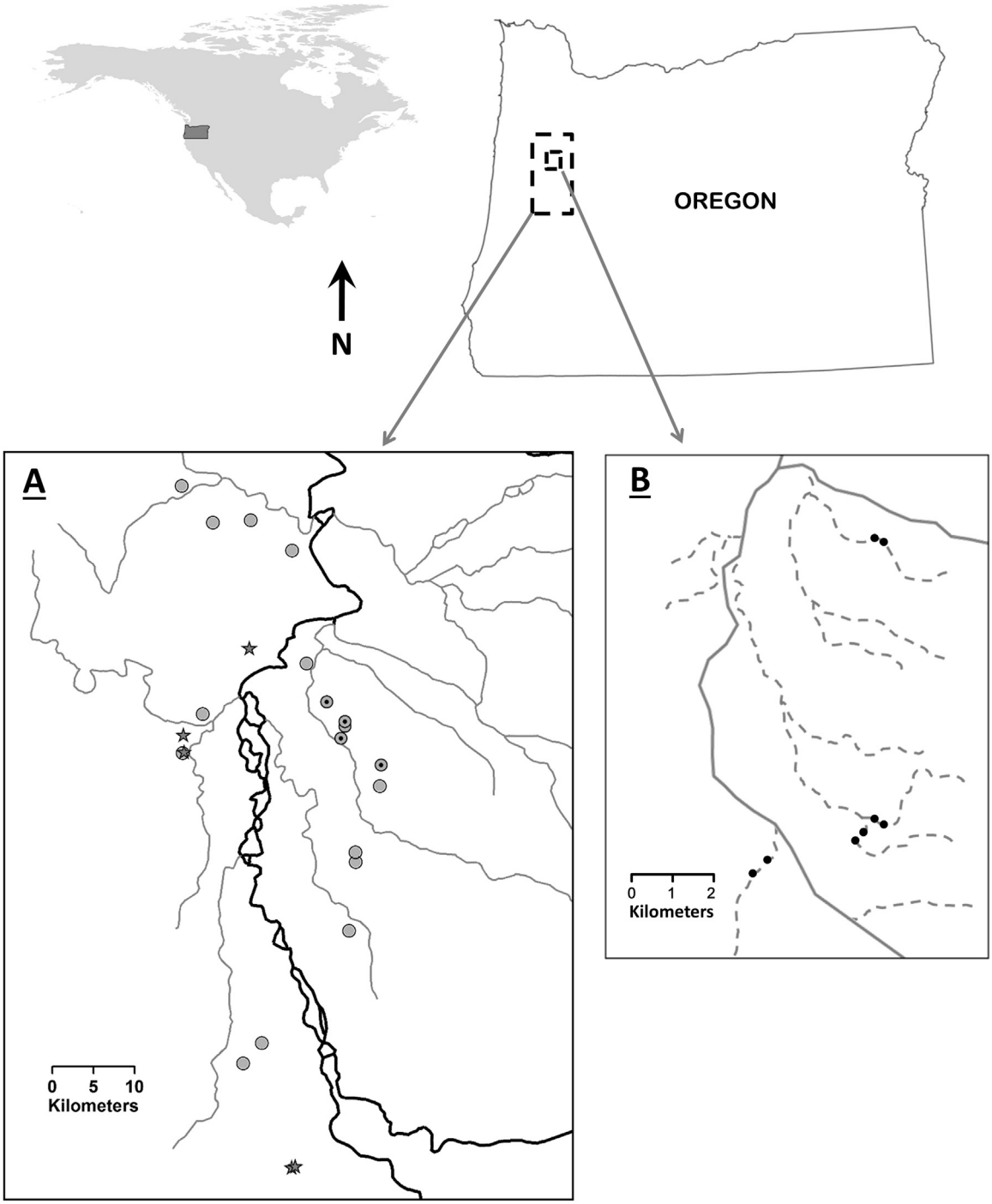

Fig. 1. Locations of southern Willamette Valley temporary stream sites sampled in 2002–03 and 2008 in western Oregon (A), and paired vegetated-non-vegetated sites sampled in 2006 (B). Black lines in map A indicate the Willamette River (flowing north). Gray lines are major Willamette River tributaries; dashed lines are temporary streams (not drawn in map A). In map A, filled gray circles indicate sites sampled in the southern valley survey (2002-03); the 5 circles with central black dots are highly agricultural sites that were resampled in 2008 to compare with least-disturbed sites. Stars indicate least-disturbed sites sampled in 2008 only.

period increases (Meyer and Meyer, 2000; Williams, 2006; Datry et al., 2014). In addition, invertebrate assemblages in streams that do not flow continuously can be quite distinct from those in permanent streams (Dieterich and Anderson, 2000; Meyer and Meyer, 2000; Williams, 2006; Storey and Quinn, 2008) with many taxa in temporary streams exhibiting traits that allow them to survive or avoid the loss of surface water (Tew, 1971; Dieterich and Anderson, 1995; Jacobi and Cary, 1996; Williams et al., 2003). In other cases, minimal differences in richness and assemblage composition between these types of streams have been found, especially when non-permanent channels being studied do not dry in all years, or where seeps, pools, or moist stream substrates remain throughout the dry period (Del Rosario and Resh, 2000; Price et al., 2003; Banks et al., 2007).

Agricultural land use has often led to degraded riparian zones, and increased water temperatures, nutrients, pesticides, and fine sediments in permanent streams (Cuffney et al., 2000; Waite and Carpenter, 2000; Anderson et al., 2003; Stone et al., 2005), but these effects are less well documented for temporary streams. In addition, in agricultural zones the hydrology tends to be altered as watercourses are often channelized, ditches are dug and, sometimes, sub-surface drainage is installed to move water quickly off the land (Boag, 1992; Armitage et al., 2003; Williams et al., 2003; Stone et al., 2005). Physico-chemical conditions associated with agriculture can be accompanied by reduced aquatic macroinvertebrate taxonomic richness, increased proportional abundances of tolerant invertebrates, and/or other indications of biological impairment (Rothrock et al., 1998; Cuffney et al., 2000; Anderson et al., 2003; Stone et al., 2005). However, impact levels may vary with the types of crops (e.g., annual row crops vs. perennial or orchard crops) or animals being reared and the intensity of production.

Effects of emergent or submerged vegetation in temporary streams are also poorly understood. Tew (1971) found in one temporary stream that the highest invertebrate density, and the greatest taxonomic richness and proportional abundance of non-insects were associated with the channel segments that had grass growing in them. This coincides with what others have reported for permanent streams, where habitats with aquatic plants that provide structural complexity and potential refuge during high flows had different assemblage composition and higher macroinvertebrate richness and density than habitats without vegetation (Cogerino et al., 1995; Armitage et al., 2001, 2003).

This study examined macroinvertebrate assemblages within a network of temporary streams flowing through agricultural lands in the southern Willamette Valley where annual and perennial grass seed were the major crops. The objectives of our research were: 1) to characterize physico-chemical conditions and watershed land use, across a broad range of sites in lowland, temporary streams; 2) to examine macroinvertebrate density, taxon richness and assemblage composition along a gradient from intense agricultural use to minimally disturbed sites; and, 3) to determine the effect of stream-bottom vegetation on macroinvertebrate density and richness in streams draining agricultural lands.

2. Materials and methods

2.1. Study location

This study was conducted in the Willamette Valley, Oregon, USA, a lowland area lying between the Coast Range and the Cascade Mountains where sediments from the Missoula floods were deposited between 15,000 and 13,000 years ago towards the end of the last ice age (Boag, 1992). The broad valley bottom has very little slope with predominantly poorly drained, clayey soils (Hulse et al., 2002; Wigington et al., 2005). The climate is characterized by cool, wet winters and warm dry summers; most of the approximately 1100 mm of annual precipitation occurs as rain. More than 95% of the precipitation falls from September through June, with about 50% in December through February (Taylor and Bartlett, 1993; Wigington et al., 2003). As a result, temporary streams in this region dry during summer and only resume flowing in October or November after the rains return. Temporary and permanent streams of the southern valley drain land that has been converted largely for agricultural use; a variety of crops are grown, but much of the land is used for grass seed production and pasture (Jackson, 1993; Klock et al., 2002; Wyss et al., 2013). According to historical accounts of Euro-American settlers in the mid-19th century, the valley was swampy during the wet season, and streams meandered and spread across the valley floor without definite channels (Boag, 1992; Taft and Haig, 2003). Since the late 19th century, however, the land has been ditched and tiled to facilitate drainage, altering the hydrology and habitat available in present-day streams. Despite alterations, stream networks expand considerably and predictably during the wet season (Wigington et al., 2005) and provide seasonal habitats for aquatic fauna (Colvin et al., 2009).

2.2. Field sampling and data collection

Sample and data collection took place in 3 stages for this study. During the 2002–2003 wet season, sites in five sub-basins of the southern Willamette Valley were sampled in conjunction with a related fish and amphibian survey (Colvin, 2006; Colvin et al., 2009). Because least disturbed sites (<55% of watershed under agriculture) were not well represented in this initial survey, in 2008 we sampled hand-picked least disturbed sites and resampled some of the same sites sampled in 2002–2003 within highly agricultural watersheds (>85% of area under agriculture) for comparison. In addition, in spring 2006 we sampled paired sites within streams in highly agricultural watersheds to test for the local effects of stream-bottom vegetation.

2.2.1. Southern valley survey, and least disturbed vs highly agricultural sites

For the southern valley survey (2002–2003), we used observations from road crossings and aerial photographs to identify a group of 59 potential sampling sites on streams likely to be dry during summer. From these, we chose 18 sampling sites in independent drainages that met the following criteria: 1) channels were dry during a scouting visit in the month of August prior to sampling, 2) watershed area sufficient to maintain water in the channels throughout the wet season (6–9 months), 3) watersheds with little urban influence and primarily draining the flattest part of the valley (Prairie Terrace level IV ecoregion) and 4) landowners granted permission for sampling (Fig. 1A).

In 2008, to identify macroinvertebrate communities likely to occur under the best conditions in these types of streams, we collected additional invertebrate samples at 5 least disturbed sites and repeated sampling in 5 of the most agriculturally influenced sites from the previous survey for comparison. The least disturbed sites had riparian vegetation and three of them were in nature preserves or park/natural areas; one stream running through private property had active wetland mitigation just beyond the riparian zone at the sampling site. Although not a criterion for selection, all least disturbed sites had evidence of beaver (*Castor canadensis*) activity nearby. The 5 highly agricultural stream sites had variable levels of riparian vegetation development (e.g., from no vegetation to corridors of shrubs or trees), and no evidence of beaver activity.

At each site, a 100 m long study section was established. In 2002–2003, water quality grab samples were collected every 2–3

weeks throughout the wet season (mid-December through early May; 7 sample dates per site). Physical habitat surveys were conducted once at each site in late April–early May 2003. Wetted widths were measured at 6 equally spaced transects perpendicular to the direction of flow in each study section. At 3 equally spaced points across these transects, depth and velocity were measured, and substrate was categorized as vegetated or non-vegetated. At non-vegetated sample points, the substrate particle size was noted. No physical habitat data or water quality samples were collected in association with samples taken in 2008.

For both years, benthic invertebrate samples consisting of 6 randomly located Surber samples (500 μm mesh netting; sampling area = 0.09 m^2 each) were collected once in late March–early April, when streams were flowing, but discharge was less variable than in winter. At each collection point, stream-bottom substrate and any rooted vegetation in the sampling frame was thoroughly disturbed and water flow pushed dislodged invertebrates and stream-bottom material into the collection net. Samples collected in this manner were preserved with 95% ethanol and were held for processing in the lab.

2.2.2. Effects of stream-bottom vegetation in highly agricultural sites

To test the effect of stream-bottom vegetation on benthic invertebrates, adjacent pairs of sites with and without extensive cover of stream-bottom vegetation (henceforward called "vegetated" and "non-vegetated" sites) were picked in 4 streams of the Calapooia River sub-basin east of the Willamette River (Fig. 1B). All sampling sites had >95% of their watersheds under agricultural use. Paired sites were approximately 200–500 m apart and within the same stream.

Each site was sampled once for invertebrates between the end of February and the beginning of April 2006. Paired sites were always sampled on the same day. Macroinvertebrate sampling was conducted in the same way as in other components of the study. In late April 2006 physical habitat surveys were conducted, but measurements were made in greater detail than those in 2003. Wetted widths were measured at 11 equally spaced transects perpendicular to the direction of flow. At 5 equally spaced points on these transects, depth and velocity were measured and substrate was characterized. Water quality grab samples were collected once at the time of habitat surveys.

2.3. Sample processing and data compilation

Water quality grab samples were collected and transported in coolers to the USDA Agricultural Research Lab on the Oregon State University campus in Corvallis, Oregon, to be analyzed for pH, conductivity, suspended sediment, nitrate-N, and orthophosphate-P. Conductivity and pH were determined using meters (Hydrolab Quanta multi-parameter water quality sensor and Orion model 811 pH meter). Suspended sediment concentration was determined by weighing the dried filtrate from a known volume of sample water. Nitrate concentration was measured by flow injection analysis (Wendt, 1999) and orthophosphate concentration was determined by flow injection analysis colorimetry (Prokopy, 1994). For the 2002–2003 southern valley survey, data were averaged across sample dates to determine the average water quality to which benthic invertebrates were exposed during the wet season. For the 2006 paired sites, water quality samples were only available from the end of April, so wet-season long averages could not be calculated.

Wetted width, depths, and velocities measured in physical habitat surveys were averaged by site for both the southern valley survey and the paired site stream-bottom vegetation test. Percent vegetated stream bottom was calculated by counting the number of points on transects where vegetation was present, then dividing by the total number of points where observations were made.

All sample site coordinates were entered into a geographic information system (GIS) (ArcGIS 8.1, Esri, Redlands, CA). Areas draining to sites were delineated using hydrologic modeling with 10 m digital elevation models (DEMs) (Oregon Geospatial Enterprise Office) and site coordinates as pour points. Watershed areas and slopes (from the highest points in watersheds to sample sites) were calculated as well as stream distances from the sampling sites to downstream permanent water. Watershed land use was determined by clipping delineated watersheds from a Willamette Valley land use/land cover GIS layer (Klock et al., 2002). Land use categories provided in the land use/land cover layer were combined and simplified to "Agriculture" (primarily perennial grass but also including annual grass, orchard, berry, and row crop fields), "Forest", "Urban", and "Other", and the proportion of each watershed in agricultural use (%Watershed Agriculture) was calculated.

The six benthic invertebrate samples collected at each site were combined to form a site composite sample. Composite samples were subsampled using a gridded sieve (Caton, 1991) with a fixed count protocol (sensu Larsen and Herlihy, 1998) and a target of identifying and enumerating 500 organisms per site. This was accomplished by taking random subsamples from each composite sample until at least 500 organisms were picked or the entire sample was processed. Aquatic invertebrates were identified to the lowest practical taxonomic resolution using dichotomous keys and taxonomic descriptions (Merritt and Cummins, 1996; Wiggins, 1996; Smith, 2001; Thorp and Covich, 2001; Stewart and Stark, 2002; Santos-Flores and Dodson, 2003). Aquatic insects were identified to genus with the exception of Simuliidae that were left at family, and Chironomidae and Ceratopogonidae that were identified to subfamily or tribe. Non-insects were identified to the finest practical level, varying from phylum (e.g., for nematodes) to genus (e.g., for amphipods and isopods). Samples were processed in the years in which they were collected, but all identifications were performed by the first author to maintain consistency among sample sets. For each site, invertebrate count data were converted to densities by dividing by the proportion of the composite sample examined and, then, dividing by the area of stream bottom sampled.

2.4. Statistical analyses

2.4.1. Southern valley survey

Invertebrate summary metrics related to whole assemblages (total density, total taxon richness) and specific types of taxa (non-insect invertebrates and insects in the disturbance sensitive orders Ephemeroptera, Plecoptera, and Trichoptera (EPT): % non-insects, %EPT, and EPT taxon richness) were calculated. In addition, we calculated values of the Hilsenhoff Biotic Index (HBI) modified for use with taxa found in Oregon aquatic environments (OWEB, 2001). HBI values are assemblage level, abundance weighted averages of taxon specific water quality tolerance values. HBI values range from 0 to 10, with low values indicating assemblages dominated by clean-water taxa intolerant of organic enrichment, and high values indicating greater assemblage pollution tolerance.

Relationships between 12 environmental variables and invertebrate density and total taxon richness were assessed with multiple regressions. The environmental variables used for these analyses were wetted width, channel depth, water velocity, % vegetated substrate, suspended sediment, nitrate-N, orthophosphate-P, conductivity, distance to downstream permanent water, watershed area, watershed slope, and watershed% agriculture. Prior to regression, variables were transformed appropriately. All possible 1, 2 and 3 variable models were run and models were

ranked by Akaike's Information Criteria modified for use with small sample sizes (AIC_c) (Burnham and Anderson, 2002). Top models were identified as all models within 2 AIC_c units of the model with the lowest AIC_c value.

Variation in assemblage composition was explored using non-metric multidimensional scaling (NMS) ordination with Bray-Curtis dissimilarity measures (PC-ORD, version 4.41, MjM Software, Gleneden Beach, Oregon). Prior to ordination, invertebrate data were log(x + 1) transformed to reduce the influence of dominant taxa on the analysis. Subsequently, data were standardized by site totals to eliminate the influence of differences in total invertebrate density among sites. Dimensionality of the ordination solution was determined using the scree test (Kruskal and Wish, 1978) to assess the point at which additional axes provided little improvement in fit (stress reduction). Site ordination coordinates were correlated with the same environmental variables used in the multiple regression analyses. Correlations between ordination coordinates and invertebrate variables (summary metrics and transformed taxa values in the ordination matrix) were also run. Directions and strengths of correlations for the most highly correlated environmental variables and invertebrate summary metrics ($r \geq 0.45$) were shown as vectors overlaid on the ordination graph.

2.4.2. Least disturbed vs. highly agricultural sites

Invertebrate metric values for least disturbed and highly agricultural sites were compared statistically using *t*-tests after appropriate transformations. Invertebrate data from this part of the study were also combined with data from the southern valley survey to give context to the previous information. Invertebrate metric medians and ranges were compared for the least disturbed, highly agricultural and southern valley survey sites. Assemblage data from sites sampled in 2003 and 2008 were combined in one matrix, log(x + 1) transformed, and standardized by site totals; assemblage composition patterns were examined using NMS ordination with Bray-Curtis dissimilarity measures.

2.4.3. Effects of stream-bottom vegetation in highly agricultural watersheds

Site physico-chemical and watershed characteristics were summarized and compared for the paired sites with and without large amounts of rooted stream-bottom vegetation. Invertebrate density and taxon richness were the invertebrate metrics used to test for effects of stream-bottom vegetation. Paired *t*-tests were run to test for all differences.

3. Results

3.1. Southern valley survey

3.1.1. Site characteristics

Median watershed area for the sites was 8.8 km^2; the largest was almost 22 km^2 (Table 1). Sites were generally several kilometers upstream of permanent water (median distance = 2.1 km), but the furthest one was 6.5 stream kilometers away from where water was present during summer. Agriculture was the major land use in all watersheds, ranging from 55% to 98% of watershed land. Perennial grass was the highest agricultural land cover in all watersheds, but the site with the lowest proportion of its watershed in agriculture also had about a third of its agricultural land in orchard/berry fields. Row crop fields were rare; they represented only 5% of the land area in the watershed where they were most abundant. Watersheds were shallowly sloping, reflecting the regional topography.

Though site specific physical measurements were made in late April-early May, we observed highly variable channel depth, wetted width, and water velocity throughout the wet season depending on rainfall. In general, these measures were highest between mid-November and late January. After February they were lower and less variable. An example of this seasonal pattern is illustrated by the hydrograph for one of the southern valley survey sites from a year in which invertebrates were not collected (2003–2004; Fig. 2). In late April–early May 2003, about a month after invertebrate sampling, median site channel depth and width were 0.30 m and 3.0 m respectively (Table 1). Median water velocity was 0.14 m/s, but site velocities ranged over an order of magnitude.

Generally, nitrate concentrations were high early in the wet season and decreased through time. Five of the 18 sites had at least one reading of nitrate-N concentration above the USEPA drinking water standard of 10 mg/L. When nitrate concentrations were averaged over the wet season, median concentration for the sites was 4.1 mg/L, but the highest nitrate concentration was almost 10 times the lowest (Table 1). Other water quality measures were also variable through time, but did not have clear seasonal trends. Averaged over the wet season, phosphate concentrations at sites ranged over approximately one order of magnitude. Conductivity and suspended sediment were less variable among sites.

Stream bottoms at all sites were predominantly hard, compressed clay. However, sites varied in the amount of rooted, stream-bottom vegetation. Plants growing in the channels

Table 1
Physico-chemical and watershed characteristics of temporary stream sites sampled in the southern Willamette Valley survey (2002–2003). Site physical measurements were taken late April or early May. Water quality data were averaged from 7 grab samples collected every 2 to 3 weeks from mid-December 2002 through early May 2003. Watershed measurements were performed with GIS. Land use data were from the Northwest Habitat Institute (Klock et al., 2002).

Variable	Median (Range)
Depth	0.30 m (0.11–0.54 m)
Velocity	0.14 m/s (0.03–0.35 m/s)
Wetted width	3.0 m (1.7–7.3 m)
Vegetated stream bottom	25% (0–83%)
NO_3-N	4.1 mg/L (0.8–7.5 mg/L)
PO_4-P	0.05 mg/L (0.01–0.11 mg/L)
Conductivity	170 μS/cm (55–244 μS/cm)
Suspended sediment	81 mg/L (68–116 mg/L)
Watershed area	8.8 km^2 (0.7–21.8 km^2)
% Watershed agriculture	86% (55–98%)
Distance to downstream permanent water	2.1 km (0.4–6.5 km)
Watershed slope	1.5% (0.1–3.3%)

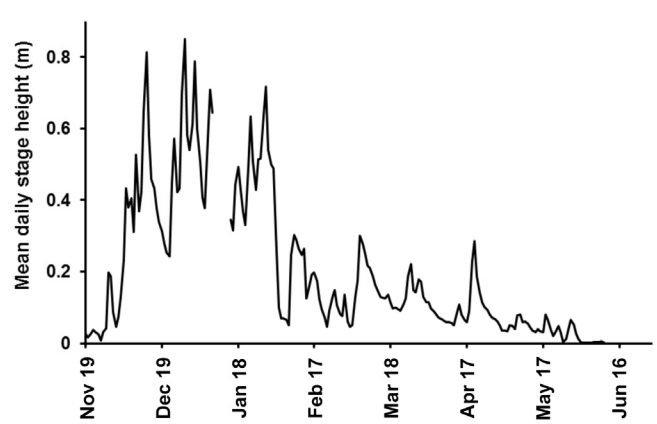

Fig. 2. Wet-season hydrograph for one of the Willamette Valley temporary stream sites sampled. Measurements from a pressure transducer in a year when invertebrate samples were not collected (2003-04).Gap in recording occurred in January while data were downloaded.

included bentgrass (*Agrostis* sp.), reed canarygrass (*Phalaris arundinacea* L.), spike rush (*Eleocharis* sp.), water plantain (*Alisma* sp.) and cattail (*Typha* sp.).

3.1.2. Benthic invertebrates

We collected a median of 18 benthic taxa per site, and a total of 49 macroinvertebrate taxa valley-wide. Non-insects collected were adults and juveniles since both are aquatic and these invertebrates don't have active, terrestrial life stages. In contrast, most insects collected were juvenile life stages, since juvenile aquatic insects in most orders (Diptera, Ephemeroptera, Plecoptera, Trichoptera, Megaloptera, and Odonata) are the ones found in water, and their winged adults are terrestrial/aerial. The exceptions among the aquatic insects are beetles and true bugs (Orders Coleoptera and Hemiptera respectively) that have aquatic juveniles; however, their winged adult forms can also be found in aquatic habitats. Median macroinvertebrate density was 2923 invertebrates per square meter, but individual site densities varied by almost 2 orders of magnitude (Table 2). These temporary stream sites had low total taxon richness (median = 18), generally dominated by non-insects (median = 88.7%, Table 2). Oligochaete and nematode worms were collected at all sites; ostracods were collected at 17 of the 18 sites (Table 3). Free-living flatworms (Turbellaria), 2 subfamilies/tribes of chironomids, and macrocrustaceans (isopods and amphipods) were found at over 80% of sites. Modified Hilsenhoff Biotic Index (HBI) scores (OWEB, 2001) between 5.5 and 7.0 indicated that these invertebrate assemblages were dominated by organisms tolerant of organic pollution and low dissolved oxygen (Table 2).

Insects in the disturbance-sensitive orders, Ephemeroptera, Plecoptera, and Trichoptera (EPT), were uncommon. Though EPT taxon richness ranged from 0 to 6, the median was only 1. Despite being collected at over 70% of the sites, *Limnephilus* spp. caddisflies were generally not abundant within sites. Even though individuals were not abundant and identifications were left at the genus level for analyses, four obvious *Limnephilus* morpho-species (based on case form and head capsule coloration) were present in the collections, indicating additional diversity within this genus.

3.1.3. Regression models

Variation in benthic invertebrate density could be explained by 4 approximately equally good multiple regression models (within 2 ΔAIC_c units of one another). Water velocity and %vegetated substrate were each in three top models. The percent of watershed in agricultural land use (abbreviated to watershed%agriculture in the model) was represented in 2 top models. The equation for the model with all of these variables was

$$\text{Log density} = 5.23 - (4.29 \times \text{velocity}) + (1.01 \times \arcsin\sqrt{\%\text{vegetated substrate}}) - (1.95 \times \arcsin\sqrt{\text{watershed}\%\text{agriculture}}).$$

None of the independent variables in this model were highly correlated with the others (all correlation coefficients: $|r| \leq 0.35$) so multicollinearity should not be a major concern and interpretation of the signs and magnitudes of the regression coefficients is straightforward (Graham, 2003). Thus, this model indicated that sites with slower water velocities, more vegetation growing in the channel, and less agriculture in the watershed had greater densities of benthic macroinvertebrates (adjusted $R^2 = 0.56$; $p < 0.01$). As individual explanatory variables, velocity and % vegetated substrate explained 34% and 32% of the variation in invertebrate density respectively; watershed%agriculture explained 7%.

Variation in taxon richness was less satisfactorily explained by the measured physico-chemical and watershed variables. Eight regression models were about equivalent (within 2 ΔAIC_c units of one another), but the best of these only explained 6% of the variation in richness ($p = 0.17$). Taxonomic richness showed little relationship to agricultural land use ($r = -0.20$; $p = 0.42$), or watershed area ($r = -0.26$; $p = 0.31$), which we thought could be a proxy for the duration of stream flow. The site with the lowest richness (9 taxa) had 73% of its watershed in agricultural land use, which was below the median agricultural land use for sites in the southern valley survey (Table 1). If this site was considered an outlier and was removed from the analysis, there was a stronger relationship between richness and watershed% agriculture ($r = -0.35$; $p = 0.07$), but not between richness and watershed area ($r = -0.19$; $p = 0.29$).

3.1.4. Community analysis

Benthic assemblage composition gradients were best represented by a three-dimensional NMS ordination. This solution accounted for 87% of the information in the original site dissimilarity matrix with a stress of 10.0. Axes 1 and 2 explained the majority of the dissimilarity, accounting for 45% and 28% respectively (Fig. 3A).

Axis 1 represented a gradient in assemblages from those with greater relative abundances of non-insects, including ostracods, nematodes, and oligochaete worms, to those with greater numbers of total and EPT taxa, presence of Simuliidae, and greater relative abundances of 2 types of chironomids (Tanypodinae and Tanytarsini). This assemblage gradient was associated with decreasing

Table 2
Benthic macroinvertebrate summary metrics for temporary stream sites from the southern Willamette Valley survey (2002-03) and least disturbed and highly agricultural sites (2008).

Variable	Southern valley survey Median (Range)	Least disturbed Median (Range)	Highly agricultural Median (Range)
Density (#/m^2)	2923 (207–14207)	5201 (3228–22471)	3465 (2393–13167)
%non-insect individuals	88.7 (31.1–99.8)	62.8 (40.9–86.1)	97.2 (77.3–100)
%EPT individuals	0.9 (0–4.9)	9.6 (1.3–24.2)	0 (0–0.7)
Total taxon richness	18 (9–26)	24 (19–27)	13 (9–15)
EPT richness	1 (0–6)	5 (2–7)	0 (0–1)
Modified Hilsenhoff Biotic Index	6.2 (5.5–7.0)	6.0 (5.2–6.5)	6.4 (5.9–7.1)

Table 3
Most common invertebrates collected in 2003 benthic samples at temporary stream sites in southern Willamette Valley. Taxa present at over 60% of the 18 survey sites are shown.

Benthic taxa	% of sites where taxa occurred	Benthic taxa	% of sites where taxa occurred
Nematoda	100	Ceratopogoninae	77.8
Oligochaeta	100	Harpacticoida	77.8
Ostracoda	94.4	*Limnephilus* spp.	72.2
Orthocladiinae	88.9	Acari	72.2
Tanytarsini	88.9	Cyclopoida	66.7
Caecidotea	88.9	*Hydra*	66.7
Crangonyx	83.3	Lymnaeidae	61.1
Turbellaria	83.3	Pisidiidae	61.1

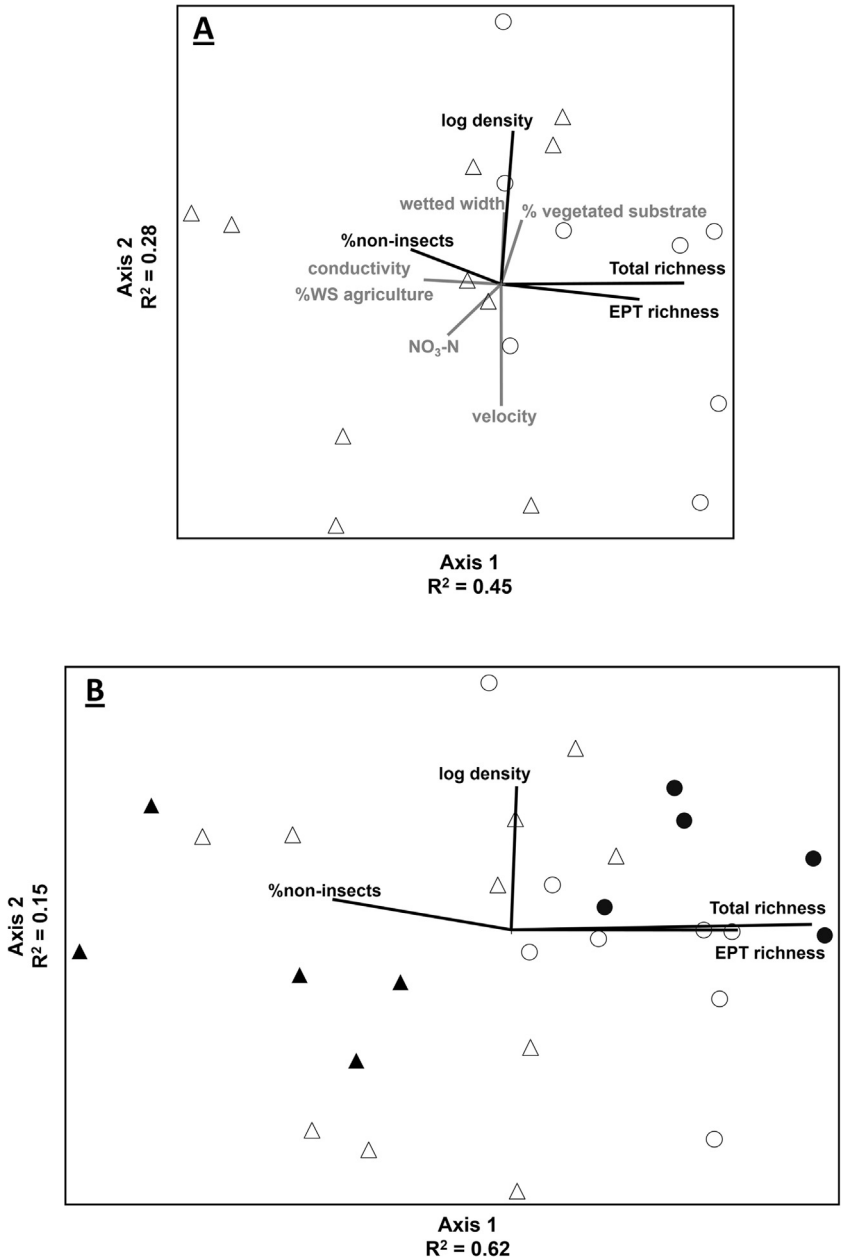

Fig. 3. NMS ordination plots of Willamette Valley temporary stream sites based on Bray-Curtis dissimilarity of benthic invertebrate assemblages. Two main axes of three-dimensional ordinations shown. A) Plot of 18 sites from the 2002–2003 southern Willamette Valley survey. Sites east (Δ) and west (O) of the Willamette River. Vectors show correlations with invertebrate metrics (black) and physico-chemical and landscape variables (gray). B) Plot of southern Willamette Valley survey sites (2002–2003) and sites from highly agricultural and least disturbed watersheds (2008). Southern valley survey sites east (Δ) and west (O) of the Willamette River, and sites from highly agricultural (▲) and least disturbed (●) watersheds.

watershed% agriculture and conductivity. Axis 1 also displayed a geographic component; most of the sites from west of the Willamette River (to the right in the ordination plot) had watersheds with somewhat less agriculture. These sites had more diverse invertebrate assemblages than sites from east of the Willamette (to the left in the ordination).

Local site conditions including stream-bottom vegetation, water velocity, and channel width were correlated with the assemblage composition gradient along Axis 2 (Fig. 3A). At sites with slower water, more vegetation growing on stream bottoms, and wider wetted channels (higher Axis 2 scores) there were higher total benthic densities and greater proportions of cyclopoid copepods, *Crangonyx* amphipods, *Caecidotea* isopods, and mites. In the opposite direction, sites with lower Axis 2 scores had higher water velocities, narrower channels, and less stream-bottom vegetation. These sites had lower benthic invertebrate densities but higher relative abundances of Orthocladiinae chironomids and *Baetis* mayflies.

Though Axis 3 also indicated a gradient in invertebrate assemblage composition, it explained little (11%) of assemblage dissimilarity, and was moderately correlated with suspended sediment ($r = 0.48$).

3.2. Least disturbed vs. highly agricultural sites

Least disturbed and highly agricultural sites sampled in 2008 were compared using the 6 invertebrate metrics listed in Table 2. Least disturbed sites had significantly greater total and EPT taxon richness (both $p < 0.01$), higher proportional abundances of EPT individuals (%EPT; $p < 0.01$) and lower proportional abundances of non-insect individuals (%non-insect; $p < 0.01$) than highly agricultural sites. However, least disturbed and highly agricultural sites did not differ in invertebrate density ($p = 0.61$), or in HBI score ($p = 0.13$).

Least disturbed and highly agricultural sites had several metric values that were at opposite ends of the ranges for sites from the 2003 southern valley survey. Least disturbed sites had some of the highest values for %EPT and taxon (total and EPT) richness, and lowest values for %non-insects (Table 2). The additional sampling in 2008 added 4 taxa to the valley-wide taxa list—3 only from least disturbed sites (a mayfly, stonefly, and caddisfly) and 1 from least disturbed and highly agricultural sites (a beetle).

HBI median values ranged from 6.0 at least disturbed sites to 6.2 and 6.4 (on a scale of 0 to 10) at other sites (Table 2). These values indicated assemblages at all sites regardless of land use were dominated by organisms tolerant of organic pollution and low dissolved oxygen (OWEB, 2001). The fact that invertebrate densities did not differ between least disturbed and highly agricultural sites indicates that the weak relationship between invertebrate density and watershed agricultural land use seen with the southern valley survey data may be spurious. Additionally even though least disturbed sites had higher %EPT than other sites, they were still dominated by other types of organisms. None of the least disturbed sites had ≥25% EPT individuals and only 2 sites had >10% EPT individuals.

A three-dimensional NMS ordination combining data from the 2003 southern valley survey and the 2008 samples (Fig. 3B) revealed assemblage gradients and correlations with invertebrate metrics that were largely the same as those seen in the single year ordination (Fig. 3A). This combined solution accounted for 91% of the information in the original site dissimilarity matrix with a stress of 10.5. Axis 1 explained 62% of the dissimilarity, and Axes 2 and 3 explained 15% and 14% respectively. Assemblages at 2008 least disturbed and highly agricultural sites were distinct from one another at opposite ends of Axis 1, while overlapping with assemblages collected in the more extensive valley-wide survey. Again there was a geographic component to this ordination because we only found least disturbed sites west of the Willamette River and the highly agricultural sites were east of the Willamette.

3.3. Effects of stream-bottom vegetation in highly agricultural sites

In the 2006 paired site comparison, rooted vegetation covered 60% to 87% of benthic substrates at vegetated sites, whereas non-vegetated sites had 0% to 7% vegetation cover. On average, sites with vegetated substrates were 10 cm shallower than corresponding non-vegetated sites within the same stream (paired *t*-test, $p < 0.01$), but paired sites did not differ in width ($p = 0.30$) or average velocity ($p = 0.78$). Average wetted widths for these sites ranged from 2 to 4.5 m and average water velocity ranged from 0.04 to 0.12 m/s. Because paired sites were close together within streams, water quality and watershed land use differences were minor. All watersheds had >95% agricultural land use.

For three streams, benthic invertebrate densities were comparatively low at non-vegetated sites (~200 to 1500 individuals per square meter) and 6–45X higher at vegetated sites (Fig. 4). These results supported findings in the southern valley survey, that local stream-bottom vegetation can be one of the primary factors affecting benthic invertebrate density. However, in a fourth stream, where the benthic invertebrate density at the non-vegetated site was more than an order of magnitude higher than in the other streams, there was little difference between vegetated and non-vegetated sites. Consequently, density differences associated with stream bottom vegetation were only marginally significant (paired *t*-test; $p = 0.08$). At these pairs of highly agricultural sites, we collected a median of 11 taxa per site. As in the 2003 survey, stream-bottom vegetation did not have a significant effect on taxon richness (paired *t*-test, $p = 0.18$).

4. Discussion

Our extensive valley-wide survey in combination with subsequent comparisons of least disturbed and highly agricultural sites and sites with and without stream-bottom vegetation revealed a suite of invertebrate taxa adapted to the seasonal nature of Willamette Valley lowland temporary streams. Many of the taxa we found in our study sites were also found in still water southern Willamette Valley seasonal wetlands (Wyss et al., 2013). Most of these taxa have resting eggs or other desiccation resistant life stages that allow them to remain dormant in place during the dry season (Williams, 2006). In addition, the aquatic insects have winged adults which allow them to colonize temporary habitats overland from nearby permanent or temporary aquatic habitats. Non-insect invertebrates, especially nematodes, oligochaetes,

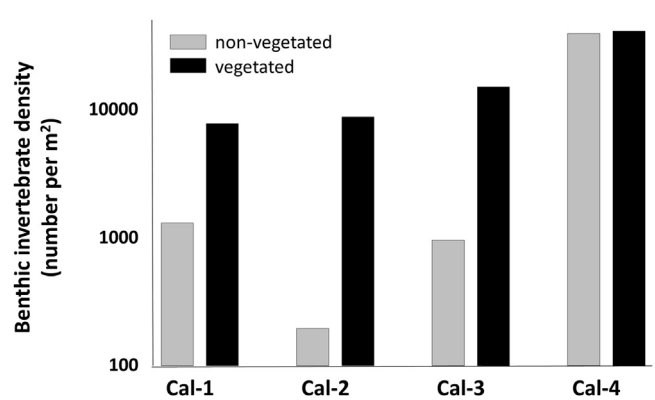

Fig. 4. Benthic invertebrate densities at paired vegetated and non-vegetated substrate sites within 4 Willamette Valley lowland temporary streams.

various crustaceans and flatworms, dominated benthic assemblages at our study sites, but in certain locations aquatic insects including those in the family Simuliidae and several EPT taxa could also be found. These invertebrates were affected by the physico-chemical conditions imposed by agricultural land use and by local stream-bottom vegetation. Though previous studies of invertebrates in non-permanent streams in agricultural landscapes examined relationships between invertebrates, local land use and physico-chemical and riparian characteristics (Aguiar et al., 2002; Davis et al., 2003; Stone et al., 2005), the broader view of our study illustrated the importance of testing for effects at both local and watershed scales.

Not surprisingly, we found that taxonomic richness in our temporary streams was considerably lower than at permanent streams in forested watersheds of western Oregon (Herlihy et al., 2005), and somewhat lower than at permanent Willamette Valley lowland streams sampled during summer (Li and Larsen, 2001). Based on the literature, we also expected that taxon richness would be related to the duration of the presence of surface water in the temporary channels we studied (Williams, 1996; Meyer and Meyer, 2000). Although we did not directly measure the duration of water presence, we expected that watershed size could be used as a surrogate for duration of the wet phase. Contrary to expectations, we did not find any relationship between taxon richness and watershed area. The temporal measure of wet phase duration may require a more local and empirical measure to detect its influence on invertebrate composition.

One of the factors that did appear to influence invertebrate taxonomic richness and assemblage composition was watershed agricultural land use. The relationship between the proportion of the watershed under agricultural use and macroinvertebrate assemblage composition was apparent in the valley-wide survey and with the addition of assemblage information from least disturbed and highly agricultural sites. On the other hand, the relationship between taxonomic richness and agriculture was not apparent across the limited range in the proportions of watershed area under agriculture in our southern valley survey, but became obvious when least disturbed sites were compared with highly agricultural sites. Because agriculture can lead to various types of habitat alteration and water quality changes (Cuffney et al., 2000; Waite and Carpenter, 2000; Anderson et al., 2003; Stone et al., 2005), and the watersheds we studied had multiple landowners with varying farming practices, we were unable to narrow down particular practices or water quality parameters that had the greatest effects on aquatic invertebrates. However, identifying specific factors associated with changes in invertebrate richness and assemblage composition and determining best management practices could be a fruitful area for future research.

We did notice that the invertebrate gradients we found that were related to agriculture also had a geographic component. The flat part of the valley (Prairie Terrace level IV ecoregion) that is more heavily used for agriculture tends to be narrower west of the Willamette River; study sites on that side of the Willamette are closer to the Valley Foothills, which have a greater proportion of their lands under forest cover (level IV ecoregion in the level III Willamette Valley ecoregion). Thus, compared to sites in the wider, flat part of the valley east of the Willamette, western sites were generally characterized by having a smaller proportion of the watershed area under agriculture, greater stream invertebrate taxonomic richness, and assemblages less dominated by non-insects. Such geographic differences in land use and invertebrate assemblage patterns have been noted in other studies at landscape scales (e.g., Aguiar et al., 2002).

Temporary stream sites were all characterized by invertebrate assemblages dominated by organisms considered highly tolerant of organic pollution and low dissolved oxygen (Modified HBI scores > 5.0; OWEB, 2001). Given the high degree of agricultural land use in many of the watersheds, it was not surprising that pollution-tolerant organisms would dominate invertebrate assemblages at most sites. However, even the invertebrate assemblages in least disturbed sites had high modified HBI scores. These high scores at least disturbed sites suggest that adaptations that enable invertebrates to complete their life cycles in temporary streams (e.g., dormant life stages, fast development, flexible life cycles, and/or good dispersal ability; Williams, 2006) could be similar to traits that allow them to inhabit streams with seasonally poor water quality. As a result, it is likely that bioassessment criteria developed for permanent streams will give misleading results for lowland temporary streams and, therefore, new criteria should be developed. The one study that contradicts this notion that temporary and permanent streams need different bioassessment criteria is from an arid region in southern California where perennial streams are rare and both temporary and perennial streams draw from the same regional pool of taxa adapted to intermittent flow (Mazor et al., 2014). The authors of this study, however, admit that their results are not likely to be applicable to wetter regions (such as Oregon) where richer regional faunas are more likely to allow the development of distinctly different assemblages in temporary and perennial streams.

Another way to look at variation in invertebrate assemblage composition across sites with differing amounts of agricultural land use is to consider the biological traits of taxonomic groups. In particular, in our study, we noted that relative abundance of non-insects was greatest in sites with high watershed agricultural land use; conversely, these sites had low abundance of aquatic insects. which were found in greater abundances in less disturbed sites. Two trait differences between aquatic insects and non-insects could account for this pattern. First, although relatively uniform, high modified HBI scores at all sampling sites indicate that invertebrate assemblages had high pollution tolerance, the average taxon specific HBI values differed between temporary stream insect and non-insect taxa. The average tolerance values for the aquatic taxa were 4.4 for insects and 6.8 for non-insects. This means, that as a group, aquatic insects are less likely to occur in temporary streams with degraded water quality than non-insects (OWEB, 2001). In addition, as indicated before, aquatic insects have winged adults, so they have greater dispersal potential than non-insects. Therefore, in watersheds with less disturbances, there may be greater source populations of aquatic insects accounting for higher colonization rates in their temporary streams.

Despite the apparent overall disturbance tolerance of invertebrate assemblages in temporary streams like the ones in this study, lowland temporary streams of the Willamette Valley are also likely important for sustaining certain specialist taxa. We collected several EPT genera (the stoneflies *Ostrocerca* and *Podmosta*, the mayflies *Ameletus* and *Anafroptilum* –formerly *Centroptilum* (Jacobus and Wiersma, 2014)- and the caddisflies in the genus *Limnephilus*) known to have species previously found in temporary streams of western Oregon (Dieterich and Anderson, 2000; Banks et al., 2007; Stewart and Anderson, 2010). Endowed with specialized adaptations to withstand or avoid the dry phase of temporary streams, these species are considered to be more abundant in, if not exclusive to, temporary habitats. In addition, we collected a little known anomopod cladoceran, *Dumontia oregonensis*, which is also known from seasonal wetlands in the Willamette Valley (Wyss et al., 2013), but was previously only reported from a few vernal pools in southern Oregon and central California (Santos-Flores and Dodson, 2003; Van Damme and Dumont, 2008). Our field sampling and laboratory subsampling procedures, and level of taxonomic identification did not allow us to develop an exhaustive list of all taxa, so there may be additional

taxa (particularly species) unique to these temporary streams that were not documented in our study.

Like us, Williams et al. (2003) found that temporary, anthropogenically modified streams in England, which they called "ditches", had lower taxon richness than perennial streams and rivers in the same region, but that they supported specialist temporary water invertebrates not found elsewhere. As a result, they considered the ditches an important aquatic habitat because of their contribution to regional biodiversity. In contrast, Datry et al. (2014) in their studies in several regions of Europe, North America, and New Zealand, did not note specialist taxa in rivers and streams without perennial flow and indicated that taxa in non-perennial lotic habitats were nested subsets of those found in perennial systems. This disagreement between studies indicates that regional or other environmental factors may determine whether or not specialist temporary water invertebrates occur in temporary streams. Further studies at broad spatial scales may help identify factors accounting for differences in current study results.

Although agricultural land use has been implicated in increasing water temperatures, nutrients, pesticides, and fine sediments in permanent streams (Rothrock et al., 1998; Cuffney et al., 2000; Waite and Carpenter, 2000; Stone et al., 2005; Anderson et al., 2006), effects of agriculture may be less for temporary streams in the southern Willamette Valley. First, because these streams dry during summer and only flow during the cooler parts of the year, removal of riparian plants that could affect summer water temperatures would not likely cause temperatures in other seasons to rise above physiological thresholds for temporary stream invertebrates. In addition, much of the cropland in the southern Willamette Valley is used for grass seed production (Jackson, 1993; Klock et al., 2002). Grasses grown for seed production complete their growth during the wet season, and seed is harvested in the dry season (Nelson et al., 2006); as a result, water withdrawal for irrigation is often unnecessary. In addition, many of the grasses grown for seed are perennial crops, so there is less soil disturbance in years after stand establishment. Consequently, these fields export less sediment and adsorbed agrochemicals to stream channels than those in which annual crops are grown. Valley-wide soil losses from water erosion in perennial rye grass fields are estimated to be 2.1 metric tons/ha/year compared to 4.5 metric tons/ha/year for annual cereal grains in this region (Steiner et al., 2006); the US national average rate of soil loss for all cropland due to water erosion is 6.1 metric tons/ha/year (USDA-NRCS, 2010). Additional soil conservation (e.g., returning chopped straw residue to fields and direct seeding/no till planting) and nutrient management strategies have been investigated, which could further protect water and habitat quality for aquatic organisms (Nelson et al., 2006; Steiner et al., 2006).

As stream-bottom vegetation management could counteract some of the effects of watershed agriculture on invertebrates, we examined the effects of different amounts of stream-bottom vegetation. Studies in permanent streams have demonstrated higher invertebrate densities and numbers of taxa associated with habitats where aquatic vegetation was present (Gregg and Rose, 1985; Armitage et al., 2001, 2003). In contrast, our results indicated that local stream-bottom vegetation affected invertebrate density but did not affect taxon richness. Though increasing benthic invertebrate density may be useful if food is limiting for native fish and amphibians in these temporary streams, stream-bottom vegetation does not seem to support more invertebrate taxa, particularly those characteristic of less agriculturally influenced sites.

Another factor to consider about stream-bottom vegetation is that we never observed high levels of stream bottom vegetation if a corridor of riparian trees was present at a study site. This was especially apparent in the four sets of paired sites used to test the effect of stream-bottom vegetation in highly agricultural watersheds, but was also noticeable in several sites in the southern valley survey. In the paired sites within the same streams, sites without riparian trees had extensive growths of stream-bottom vegetation, while just a few hundred meters away sites with riparian trees had very little stream-bottom vegetation. Perhaps shading from the riparian canopy prevents dense growth of stream-bottom plants. However, in the Willamette Valley, areas with riparian trees and shrubs support more wintering birds and bird species than those without woody vegetation (McComb et al., 2005), and areas with trees adjacent to bodies of water are important for amphibians that make extensive use of upland habitat (Pearl et al., 2005). Thus, it seems there may be a need to find a balance between encouraging stream-bottom vegetation for increasing invertebrate densities and maintaining habitat conditions that benefit several native vertebrate species in the riparian habitats.

Another option to enhance invertebrate assemblages could be to take some agricultural land out of production. Our results show that even modest reduction in the percentage of watershed land in agriculture could be helpful, because at sites with approximately 75% of watershed area under agricultural use, the invertebrate assemblages were similar to those at least disturbed sites. Currently, many low-lying areas in the southern Willamette Valley are farmed despite the fact that they remain wet for extended periods and consequently have lower crop yields (J.J. Steiner, USDA-ARS, personal communication). These lands with lower crop production could potentially be set aside with the aid of federal conservation programs to maximize biodiversity and minimize effects on farm incomes.

5. Conclusions

Our study found a range of invertebrate assemblages that included dominant non-insect groups characterized by drought-resistant stages that survive the dry period on site, and insects that may have resistant stages for on-site survival, but also have winged adult stages to disperse to and from other habitats. Because our results suggest that invertebrates, integral to wet-season food webs, are affected by agricultural land use, these aquatic communities should be integrated into conservation planning and agricultural management. Studies of other temporary streams in agriculturally rich regions will further our understanding of how to balance the conservation of stream biota within the context of vital agricultural landscapes.

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