MAPPING DYNAMIC EXPOSURE: CONSTRUCTING GIS MODELS OF
SPATIOTEMPORAL HETEROGENEITY IN ARTIFICIAL STREAM SYSTEMS

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ABSTRACT

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In flowing environments, the degree of turbulent flow determines the movement and distribution of chemicals. Variation in flow alters the patchiness of toxicant plumes within a stream ecosystem. This patchiness translates into variability in exposure pulses for organisms encountering the toxic plume. Throughout a stream, the processes that give rise to chemical plume structure will vary as a function of local flow characteristics. This research examines the influence of toxicant mode of entry and stream flow velocity on the spatiotemporal patterning of exposure. Two introduction treatments were evaluated: one mimicking groundwater and the other mimicking runoff. The influence of flow regime was examined through the comparison of models constructed under two stream flow velocities. Concentrations of a tracer molecule were recorded using an electrochemical monitoring system. From these localized, direct measurements, geographic information systems (GIS) were used to model exposure throughout the stream. Conceptualizing exposure as a series of toxicant pulses, exposure can be defined using a variety of chemical peak characteristics. Three-dimensional, layered maps were constructed defining exposure as the integrated area of toxicant peaks, the magnitude of peaks, and peak frequency. Differences in the spatial and temporal patterning of exposure were apparent both within treatments and between treatments. No two definitions of exposure yielded the same exposure distributions for any treatment. These models demonstrate that distribution of chemical exposure throughout a stream ecosystem is linked to both toxicant mode of introduction and

stream hydrodynamics. Furthermore, these results demonstrate that optimal exposure modeling relies on first defining exposure.

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INTRODUCTION

The introduction of anthropogenic pollutants to aquatic ecosystems has emerged as an issue of increasing concern in recent decades (Abel 2002; Amiard-Triquet et al. 2015). Chemical introduction to natural bodies of water occurs, both deliberately and unintentionally, as a consequence of industry, energy production, agriculture, forestry, wastewater management, and a variety of other human activities (Herlihy et al. 1990; Abel 2002; Collier et al. 2016; Stumpf et al. 2016). While anthropogenic toxicants pose a threat to nearly all types of aquatic ecosystems, stream habitats are particularly susceptible to these inputs, as pollutants are able to distribute downstream rapidly (Collier et al. 2016). A ten-year survey by the USGS National Water-Quality-Assessment (NAWQA) Program reported that 56% of sampled streams with developed watersheds contained pesticide concentrations above those considered safe for aquatic wildlife (Gilliom 2007). In a national survey conducted by the U.S. Environmental Protection Agency, measurable concentrations of polychlorinated biphenyl (PCBs) were found in fish tissues for 93.5% of sampled river sites (Batt et al. 2017). The high prevalence of anthropogenic contamination in stream environments makes the direct monitoring of all ecosystems of concern impractical, if not impossible.

In lieu of direct monitoring, effective management of natural water resources requires the use of ecotoxicological modeling (Palmer 2001; Benedini and Tsakiris 2013; Gu 2014). Appropriate modeling, as well as resultant risk assessment, calls for an understanding of how chemicals move through natural stream environments. A deeper understanding of toxicant plume movement will allow for more accurate prediction of organismal exposure (Handy 1994; Ashauer et al. 2006). As such, the incorporation of the hydrodynamics of chemical entry and transport in stream habitats will allow for improved modeling of exposure patterns.

Contaminants are introduced to stream environments through a multitude of pathways. Commonly, anthropogenic chemicals are carried into streams by contaminated groundwater or surface runoff. A wide range of land use, biotic, geological, and hydrodynamic factors influence the occurrence of contaminated inputs from groundwater and runoff (Sophocleous 2002; Loperfido et al. 2014). Groundwater enters a stream through the streambed, forming the baseflow of the stream (Bear and Cheng 2010). Toxicants from either surface water or soil can be introduced into groundwater aquifers as water percolates vertically downwards through the soil profile (Reichenberger 2007). Susceptibility of groundwater to contamination is largely influenced by the shallowness of the water table, precipitation rates, and soil texture, as these factors contribute to the ease of surface water percolation to aquifers (Kreuzinger et al. 2004; Reichenberger 2007; Al-Rawabdeh et al. 2014). Groundwater sources are also likely to contain chemicals utilized in wastewater treatment in particularly dry regions, where treated water is used to artificially replenish groundwater reserves (Kreuzinger et al. 2004). In contrast, runoff carries chemicals from neighboring terrestrial environments into the surface waters of aquatic environments. Runoff introduction is typically tied to precipitation events and most often results in the diffuse, non-point source introduction of herbicides, insecticides, and chemicals used in wastewater treatment (Wang et al. 2014; Amiard-Triquet et al. 2015).

Because chemicals introduced via runoff and groundwater enter streams at different points in the water column, they will encounter vastly different flow regimes (Vogel 1994; Nikora 2010; Steele et al. 2018). As such, these two modes of entry are likely to result in different patterns of chemical transport (Edwards and Moore 2014; Lahman and Moore 2015). In fluids, the transport of chemicals is primarily a product of two distinct processes: advection and diffusion. Advection refers to large-scale transport of chemicals by flow and facilitates

patchiness within chemical plumes. By comparison, diffusion gradually acts to eliminate spatial gradients in chemical concentrations (Denny 1993; Moore and Crimaldi 2004). In water, molecular diffusion is a slow process, slow enough as to become irrelevant at the scale of aquatic macroinvertebrates and larger organisms. At these scales, advective stirring is the major physical process that determines spatial and temporal variation in the structure of chemical plume (Sanford 1997; Moore and Crimaldi 2004; Webster and Weissburg 2009).

Having established that advective forces dominate the transport of chemicals in stream environments, the transport of toxicants can be described by characterizing flow. The character of flow within a stream can generally be described using the Reynolds number, which reflects the relative influence of inertial and viscous forces within a fluid (Denny 1993; Vogel 1994). Increasing Reynolds numbers are associated with the dominance of inertial forces, reflecting more turbulent flow. Variation in the degree of turbulent flow gives rise to variation in patchiness in the structure of chemical plumes. This patchiness translates into variability in exposure pulses for any organisms that encounter the plume (Wolf et al. 2009; Edwards and Moore 2014). Any factors that affect flow dynamics throughout a stream habitat will alter the patchiness of chemical plumes, ultimately shaping exposure (Ludington and Moore 2017). Near fluid-surface interfaces, a boundary layer forms between the inert fluid at the interface and free stream velocity (Vogel 1994). In stream ecosystems, the boundary layer above the substrate creates lesser degrees of turbulent flow, resulting in lessened patchiness in chemical plumes near the substrate. Further from the boundary layer, at free stream velocity, increased turbulence facilitates greater variation between toxicant peaks (Zimmer-Faust et al. 1988; Moore et al. 1994; Finelli et al. 2000). Stream environments are comprised of a network of distinct microhabitats; as such, flowing systems are enormously heterogeneous in both space and time (Thorp et al. 2006;

Schletterer et al. 2010). Heterogeneity in across stream environments lead to variability in the patterning of exposure for organisms living in fluid. However, most conventional approaches to exposure modeling do not reflect this variability.

Historically, evaluations of toxicity have relied on a static exposure paradigm. Under a static exposure paradigm, organisms are exposed to a chemical at a consistent concentration over a given time-period (Watanabe et al. 2018; Gordon et al. 2012). However, organisms in natural systems rarely encounter constant and consistent exposure to toxicants (Ashauer and Escher 2010; Edwards and Moore 2014). Pulsed exposure paradigms have introduced temporal variation into models of toxicant exposure. Pulsed exposure paradigms represent a significant improvement in exposure modeling, as they reflect the large-scale episodic nature of toxicant exposure in natural environment (Butcher et al. 2006; Watanabe et al. 2018). However, these pulsed models of exposure cannot adequately account for discrepancies in the concentration, duration, and frequency of exposure peaks that emerge at fine scales in natural systems (Milne et al. 2000; Ludington and Moore 2017; Neal and Moore 2017). The significance of the variability in toxicant concentrations for aquatic organisms is two-fold. Firstly, in turbulent environments, organisms may experience peak concentrations of a toxicant that are much higher than the average concentrations predicted utilizing Gaussian models of exposure (Moore et al. 1992; Ludington and Moore 2017). Secondly, fluctuation in exposure in-and-of itself has been demonstrated to influence the extent of a toxicant's detrimental impacts when compared to consistent exposure to the same mean toxicant concentration (Gordon et al. 2012; Neal and Moore 2017). Furthermore, differences in the toxicant mode of entry have been directly tied to the consequences of exposure (Lahman and Moore 2015; Steele et al. 2018).

Turbulent dispersion creates a complex landscape of toxicant distribution (Atema 1996). This landscape consists of constantly evolving peaks. To model this type of dynamic exposure in streams appropriately, the interaction between the method of introduction of contaminants and the underlying turbulence of a flowing environment needs to be understood. In addition, considering only the mean exposure concentrations at specific locations is insufficient in describing the toxicity of contaminants. Peak characteristics such as magnitude, duration, frequency, and intermittency allow for a fuller understanding of the dynamic nature of chemical exposure in turbulent, flowing systems (Gordon et al. 2012; Ludington and Moore 2017). Using experimental systems to model stream characteristics that may generate turbulence, and thereby patchiness in toxicant distribution, we can gain insight into how toxicant impacts may vary across and between stream environments (Harrigan and Moore 2017). The purpose of this study is to create spatial models of the distribution of exposure, defined as specific peak characteristics, throughout a stream environment. These maps can be utilized to compare patterning of exposure in streams of different flow velocities, taking both mode of toxicant introduction and organism position in the water column (benthic, mid-water column, or surface) into consideration.

METHODS

Experimental Design

This experiment utilized a 2 x 2 fully factorial design to assess the contributions of mode of chemical introduction and stream flow velocity to chemical plume structure. Two chemical introduction treatments were evaluated: the first mimicking groundwater introduction and the second mimicking runoff introduction. The influence of flow regime was examined through the comparison of models constructed under two stream flow velocities. Ultimately, this experimental design resulted in the examination of four treatments:

- Groundwater Introduction x Low Stream Flow Velocity
- Groundwater Introduction x High Stream Flow Velocity
- Runoff Introduction x Low Stream Flow Velocity
- Runoff Introduction x High Stream Flow Velocity

For all treatments, a tracer solution was introduced into the stream via a constant head tank to mimic a toxicant introduction event.

Flume Construction

An artificial stream was constructed at the University of Michigan Biological Station Stream Research Facility in Pellston, MI. The flume (17.4 x 0.98 x 0.40 m: L x W x H) was constructed using cinder blocks lined with 0.1 mm thick plastic sheeting (Fig. 1). The bottom of the flume was lined with approximately 3 cm of a combined sand-gravel substrate to simulate the bottom of a natural stream. Water from the East Branch of the Maple River was continuously pumped into the flume to create a depth of approximately 28 cm, which was held constant across all treatments. Nylon mesh was placed over the stream inflow to filter out any large particulate

matter that could influence flow within the stream (Ludington and Moore 2017). The flume contained two primary sections: a mixing section (2.1 m) and a working section (0.75 m).

After filtration, water flowed into the mixing section of the flume. In the mixing section, water passed through a collimator to minimize the larger and more chaotic elements of turbulence resulting from water entry into the flume. The collimator consisted of a series of three pieces of plastic egg crating (1.7 x 1.7 cm openings). The outer two layers of egg crating were wrapped in a wire mesh (0.14 x 0.16 cm mesh). After passing through the collimator, water flowed 6.9 m downstream to enter the working section of the flume.

The working section of the stream contained both the delivery system (0.15 m) and the sampling area (0.3 m). In the delivery system area, a dopamine tracer stimulus was introduced as either groundwater or runoff, as dictated by treatment (Fig. 2). Water flowed 30 cm downstream of the delivery system to enter the sampling area. The sampling area of the flume was located 7.35 m from the outflow area of the flume, to avoid the influence of end effects on flow (Fig. 3).

Water exited the stream at an outflow downstream of these sections and was subsequently drained back into the Maple River. This ensured the continual introduction of fresh water into the artificial stream system.

Regulation of Stream Velocity

Stream velocity, as specified by treatment, was controlled at the flume's inflow and outflow. At the flume's inflow, the natural river water could be supplemented by additional well-water inputs to obtain higher stream flow velocities. At the outflow, stream velocity could be controlled with the placement of boards to obstruct water leaving the stream. Obstructions with varying degrees of porousness were used to manipulate stream flow velocity. Velocity was held constant for the duration of each trial.

Under low stream velocity treatments, stream flow velocity was held constant at 2.02 ± 0.23 cm/s. Under high stream velocity treatments, stream flow velocity was held constant at 9.56 ± 0.73 cm/s. Stream flow velocity was measured using a Hach Flow Meter (Hach FH950, Loveland, Colorado), which was placed 14 cm above the substrate in the center of the stream. This placement was chosen to measure free stream velocity without the influence of the boundary layer or wall effects. Velocity under each treatment was measured at ten second intervals for a period of five minutes.

Stimulus Delivery

Dopamine-fluorescein tracer solution (0.113 g/L) was pumped from a constant head tank through a standard 0.476 cm diameter Tygon tube. Tracer solution entered the stream through a series of four plastic nozzles placed in the center of the stream 30 cm upstream of the sampling grid (Fig. 3). These nozzles allowed for the precise control of tracer solution flow, which was held constant at a rate of 60 ml per minute across all treatments. For groundwater introduction treatments, the series of nozzles was buried approximately 2 cm deep in the substrate. For runoff introduction treatments, nozzles were suspended 10 cm above the water's surface.

Sampling Protocol

Concentrations of dopamine were monitored using an EmStat3+ Blue electrochemical monitoring system (PalmSens, Houten, Netherlands). Dopamine has a similar molecular weight (MW: 153.18) to many prevalent organic toxicants, and the diffusion coefficient of dopamine (2×10^{-5} cm²/s) is within an order of magnitude of the diffusion coefficient of such toxicants (Harrigan and Moore 2017). Considering that the influence of diffusion is relatively minor, the diffusion coefficient of dopamine is sufficiently similar to those of organic toxicants that dopamine can be expected to mimic their distribution closely.

The EmStat measures electrical currents resulting from oxidation-reduction reactions as tracer molecules encounter an electrode. The microelectrodes utilized in this experiment were built in-house. The electrodes in their entirety are approximately 5 cm long and are composed of three carbon fibers. These carbon fibers measure 30 μm in diameter and approximately 1.5 cm in length. Together with carbon paste, the three carbon fibers are encased in a borosilicate glass housing, with fibers protruding approximately 1.5 cm out of one side of the housing. Copper wire extends out of the opposite side of the electrode; this wire was connected to the EmStat using alligator clips. These microelectrodes allow for sampling of chemical concentrations at temporal and spatial scales pertinent to aquatic macroinvertebrates (Harrigan and Moore 2017; Schneider et al. 1998; Moore et al. 1989).

Before in-stream sampling could be completed, the sensitivity of each individual electrode had to be determined. To capture a range in frequency observable in both laminar and turbulent flow, the electrode sampled at 20 Hz. Calibrations were completed by introducing a sequence of three known dopamine concentrations into water sourced from the same stream as was used in the flume and monitoring the resultant electrical current at the electrode. The response curves exhibited a linear relationship and could later be used to convert measurements of electrical current taken within the stream into corresponding dopamine concentrations.

After electrode calibration, chemical concentration profiles were taken at 20 sampling sites following a 5 x 4 centroid grid (30 cm x 30 cm) in the sampling area (Fig. 3). For each sampling site, measurements were taken at three heights within the water column: 3 cm, 14 cm, and 25 cm above the flume's substrate (Fig. 3). Tracer molecule concentrations were monitored at each of the 60 locations over a period of 5 minutes each. This allowed for a representative sample of fluctuations in dopamine concentrations at that location in the stream (Moore et al.

1994). Based on signal sampling theory, a 5 minute sample taken at 20 Hz is enough to capture the entire range of chemical fluctuations within this system (Box et al. 2015). This sampling protocol produced a sequence of 6,000 measurements of oxidation-reduction reactions occurring at the electrode for each location in the sampling grid under a given treatment.

Peak Analysis

Concentration profiles were exported out of the EmStat's native PStouch software into Microsoft Excel Version 1811. In Excel, the initial measurements of oxidation-reduction reactions at the electrode were converted into dopamine concentrations using the previously completed calibration curve for that electrode. This procedure follows established techniques for the analysis of spatiotemporal fluctuations in chemical signals using microelectrodes (Moore et al. 1989; Schneider et al. 1998; Wolf et al. 2009; Harrigan and Moore 2017).

Peaks within each trial were identified using the Peak Analysis extension of Origin Version 2018 B 64-bit software. Peaks were defined as any increase in concentration reaching 2.5 times average background, with peaks ending once concentration reached 20% of maximum peak height. Peak characteristics that are likely correlated with the degree of impact of an exposure event on an exposed organism were then analyzed. These characteristics of interest included the sum of the integrated area under the peaks, the magnitude of peaks, and the frequency of peaks observed at a given sampling site. Frequency was defined as the number of peaks recorded during the 5-minute sampling interval (Fig. 4). Magnitude was defined as the highest concentration of the tracer molecule recorded during each peak (Fig. 4). An approximation of total exposure at a given sampling site was obtained by integrating and summing the area under each of the peaks.

Map Construction

Data describing each of the peak characteristics were imported into ArcGIS Desktop 10.5 (Esri, Redlands, CA). Ordinary Kriging was used to interpolate between sampling points within each layer, creating an approximation of the distribution of each peak characteristic under each treatment. Kriging was selected as the most appropriate interpolation method for this study because estimations of values at unsampled locations incorporate both distance from sampled points and relative positions to sample points (Coulibaly and Becker 2007). The use of a geostatistical approach to interpolation allows for model sensitivity to directional relationships, which are not reflected with the use of deterministic methods such as the Inverse Distance Weighted Method (IDW) (Coulibaly and Becker 2007). For maps of peak magnitude, the mean value of peak maximums observed was used to create a single value for each sampling point. Because peak area (summed) and peak frequency produced only one value per sampling point, no such process was needed in creating maps of their distributions.

The result of this interpolation process was a map of each peak characteristic for all sampling grid layers under each treatment. A raster layer was then created from each of these maps, and the raster layer was imported into ArcScene 10.5 (Esri, Redlands, CA). Once the three raster layers for a specific treatment and peak characteristic were imported into ArcScene, the appropriate “layer offset” was added to each layer. The layer offset imposes a constant elevation value for each layer, allowing them to be stacked vertically while maintaining scale. Because each of the maps were created using a coordinate grid in cm, the appropriate elevations were 3, 14, and 25 grid units, respectively.

Using these three-dimensional maps, areas of the stream in which organisms are likely to experience the most severe impacts of anthropogenic chemical introduction were

identified using a red-blue color scheme. To normalize across all treatments, color breaks were assigned by percentile within the range of all observed values of a peak characteristic. Breaks were assigned at 5% intervals. Red represented the highest levels of exposure, falling into the uppermost percentiles of a given peak characteristic. Blue was chosen for the lowest levels of exposure. Root Means Square (RMS) values were extracted from each map layer to estimate error (Table 5). The partial sill and range of each layer were also extracted in order to evaluate the distance at which points within a layer ceased to be spatially autocorrelated (Table 6).

RESULTS

Groundwater Introduction x Low Stream Flow Velocity

In groundwater introduction treatments at low stream flow velocities, the highest levels of exposure across all definitions (peak area, peak magnitude, and peak frequency) were observed primarily in the layer 3 cm above the substrate (Table 1; Fig. 6). For both peak area and peak magnitude, levels of exposure falling into the uppermost quartile of observed values for each definition comprised approximately half of the layer's total area. By comparison, when defining exposure as peak frequency, levels of exposure were not as extreme in this layer, with the majority of the layer experiencing numbers of peaks falling into the bottom two quartiles.

Defining exposure as the peak area, values falling into the upper two quartiles of exposure accounted for less than 1% of the layer 14 cm above the substrate. The vast majority of this layer experienced levels of exposure falling into the bottom-most quartile of observed peak areas. Similarly, no levels of exposure falling into the upper two quartiles were observed in the layer 14 cm above the substrate when defining exposure as the peak magnitude. However, this layer was considered to undergo higher levels of exposure when defining exposure as peak magnitude, as compared to peak area. The peak frequency was the only exposure definition under which values falling within the upper two quartiles of exposure were observed in this layer.

Levels of exposure in the uppermost quartile were not observed in the layer 25 cm above the substrate using peak area or peak frequency as definitions of exposure. Levels of exposure in the third quartile were not observed in this layer when defining exposure as the peak area and were observed in less than 1% of the layer when defining exposure as the peak frequency. In

contrast, levels of exposure falling into the upper two quartiles accounted for the vast majority of the layer when exposure was defined as the peak magnitude.

Groundwater Introduction x High Stream Flow Velocity

In groundwater introduction treatments at high stream flow velocities, the highest levels of exposure (falling in the uppermost quartile) defined as peak area were observed only in the layer 3 cm above the substrate (Table 2; Fig. 7). Defining exposure as the peak magnitude, an even larger proportion of this layer experienced levels of exposure falling into the uppermost quartile. In contrast, levels of exposure falling into the uppermost quartile were not observed in this layer when exposure was defined as the peak frequency. Most of the layer experienced levels of exposure falling into the lower two quartiles when exposure was defined as the peak frequency.

Defining exposure as the peak area, values falling into the middle two quartiles of exposure accounted for the largest proportion of the layer 14 cm above the substrate. Over half of the layer experienced levels of exposure falling into the third quartile, and about a third of the layer experienced levels of exposure falling into the second quartile. By contrast, levels of exposure in the highest quartile accounted for over 86% of the layer. When exposure was defined as the peak frequency, the layer was observed as having lower levels of exposure than under either peak area or peak magnitude definitions. Levels of exposure in the lowest quartile accounted for more than half of the layer's total area, with numbers of peaks falling into the second quartile accounting for the next-largest proportion of the layer.

Similarly to what was observed in the layer 14 cm above the substrate, the vast majority of the layer 25 cm above the substrate experienced levels of exposure in the middle two quartiles when exposure was defined as the peak area. No levels of exposure in the uppermost quartile of

observed peak areas were observed in this layer, and levels of exposure falling into the lowest percentile accounted for less than 1% of the layer's total area. However, when defining exposure as the peak magnitude, levels of exposure in the uppermost quartile accounted for approximately half of the layer 25 cm above the substrate. Defining exposure as the peak magnitude, the remainder of area in the layer experienced intermediate levels of exposure, with levels of exposure in the lowest quartile comprising about 5% of the layer. Overall, peak frequency identified the greatest proportion of this layer as experiencing low levels of exposure, with over 72% of the layer experiencing numbers of peaks falling into the bottom quartile. Peak frequency was the only definition of exposure under which the largest proportion of high exposure (falling in the uppermost quartile) in any layer was observed in the layer 25 cm above the substrate.

Runoff Introduction x Low Stream Flow Velocity

In runoff treatments at low flow velocities, levels of exposure falling into the third quartile were observed in over 76% of the layer 3 cm above the substrate (Table 3; Fig. 8). Defining exposure as the peak area, this layer had the largest proportion of area experiencing levels of exposure in the upper two quartiles. These levels of exposure are lower than those observed 3 cm above the substrate when exposure was defined as the peak magnitude, but higher than those observed when exposure was defined as the peak frequency. When exposure was defined as the peak magnitude, levels of exposure in the uppermost quartile accounted for over 95% of the layer's total area. By contrast, levels of exposure falling into the upper two quartiles were not observed in this layer when exposure was defined as the peak frequency.

Defining exposure as the peak area, the vast majority of the layer 14 cm above the substrate experienced levels of exposure falling into the middle two quartiles. These intermediate levels of exposure were observed in more than 98% of the layer's total area. When exposure was

defined as the peak magnitude, more than 95% of this layer experienced levels of exposure in the uppermost quartile; no intermediate levels of exposure were observed. When exposure was defined as the peak frequency, levels of exposure were more evenly distributed across quartiles in this layer. Levels of exposure in the bottom quartile comprised well over half of the layer when exposure was defined as the peak frequency.

Of all layers in runoff treatments at low stream velocities, the layer 25 cm above the substrate had the highest proportion of peak areas falling into the highest quartile. These relatively high levels of exposure accounted for over half of the layer. Levels of exposure in the uppermost quartile accounted for an even larger proportion of this layer when exposure was defined as the peak magnitude, comprising more than 85 percent of the layer. As was observed in the layer 14 cm above the substrate, degrees of exposure in this layer were most evenly distributed across the quartiles when exposure was defined as the peak frequency.

Runoff Introduction x High Stream Flow Velocity

In runoff treatments at high flow velocities, no peak areas falling in the two upper quartiles were observed in the layer 3 cm above the substrate (Table 4; Fig. 9). The vast majority of the layer was observed to demonstrate levels of exposure falling into the second quartile, with exposure levels in the second quartile accounting for over 99% of the total layer area. By contrast, higher levels of exposure were observed much more frequently in this layer when exposure was defined as the peak magnitude. Defining exposure as the peak magnitude, levels of exposure falling into the uppermost quartile accounted for 59% of the layer, with the remainder of the layer experiencing peak magnitudes falling into the second and third quartiles. When exposure was defined as the peak frequency, levels of exposure were more evenly distributed

across quartiles in this layer. Levels of exposure in the bottom quartile comprised the largest proportion of the layer when exposure was defined as the peak frequency.

Defining exposure as the peak area, the entirety of the layer 14 cm above the substrate experienced levels of exposure falling into the lower two quartiles. These relatively low levels of exposure also comprised the majority of the layer when exposure was defined as the peak magnitude, accounting for approximately 90% of the layer. Of the three measures of exposure, levels of exposure falling in the uppermost quartile were observed most commonly in this layer when exposure was defined as the peak frequency, comprising nearly half of the layer.

Levels of exposure in the uppermost quartile of observed peak areas accounted for more than half of the layer 25 cm above the substrate. While the relatively high levels of exposure comprised the majority of the layer when exposure was defined as the peak area, they were observed even more frequently when exposure was defined as the peak magnitude. Levels of exposure in the uppermost quartile of observed peak areas accounted for about 71% of the area in this layer. In contrast, a majority of the layer experienced relatively low levels of exposure when exposure was defined as the peak frequency, with values falling into the lowest quartile being observed in more than half of the layer.

DISCUSSION

Across all treatments and measures of exposure, the distribution of exposure throughout the stream was observed to be heterogeneous. Differences in the spatial and temporal patterning of exposure were apparent both within treatments (at different levels in the water column) and between treatments. Spatial and temporal heterogeneity is to be expected given the degree of turbulence in the stream system, and the structure of this heterogeneity was shown to be dependent on both mode of entry and stream flow velocity. Mode of toxicant entry into the stream was shown to interact with stream flow velocity to create distinct distributions of exposure. Examining the exposure distributions created under different degrees of turbulence, the areas within the stream that were identified as experiencing the highest levels of exposure changed dramatically depending on the peak characteristic that was used to define exposure.

Variation Across Treatments

The models constructed in this study illustrate that the point at which a toxicant enters a stream influences the patterning of the chemical plume moving through the stream environment (Wolf et al. 2004; Steele et al. 2018). Overall, chemicals entering the stream via groundwater introduction were more likely to remain within the boundary layer (Fig. 6; Fig. 7). By comparison, when chemicals entered the stream via runoff introduction, they were more prone to distribute throughout the water column (Fig. 8; Fig. 9) (Vogel 1994; Nikora 2010). In groundwater introduction treatments, the layer nearest the substrate consistently experienced the largest peaks with the highest maximums. The same pattern was not observed under runoff treatments, where high levels of exposure were commonly observed in the upper layers of the water column as well. As chemical transport is primarily accomplished by advection at the scales sampled in this study, this trend likely reflects the initial degree of turbulence encountered by the

chemical solution (Denny 1993; Moore and Crimaldi 2004). Under runoff introduction treatments, chemicals immediately encounter turbulence as they enter the flume at free stream velocity, whereas chemicals in groundwater treatments initially encounter relatively laminar conditions in the boundary layer near the substrate (Vogel 1994; Nikora 2010; Steele et al. 2018).

In addition to the influence of the mode of introduction, exposure distributions were influenced by stream flow velocity. Turbulence resulting from higher flow velocity appeared to facilitate increased rates of exchange between the boundary layer and the free stream. In groundwater treatments at low flow velocities, the largest peak areas and peak frequencies were contained almost exclusively within the boundary layer (Fig. 6). Under both definitions of exposure, the vast majority of the upper layers of the water column experienced levels of exposure falling into the lowest quartiles. Only when exposure was defined as the peak magnitude were relatively high levels of exposure observed outside of the boundary layer (Fig. 6). Overall, intermediate and high levels of exposure following groundwater introduction were much more prevalent in the upper layers of the water column at high flow velocities (Fig. 7). In contrast, at low stream flow velocities, runoff introductions frequently led to a large proportion of all three layers experiencing the high peak areas and maximum magnitudes (Fig. 8). Opposite to what was observed in groundwater treatments, intermediate and high peak sizes were observed less commonly outside the layer of introduction at high flow velocities following runoff introduction (Fig. 9). One explanation for this trend may be that, owing to the increased stream flow, tracer solution was carried further downstream than the sampling grid before turbulence distributed the tracer into the lower layers (Crimaldi et al. 2002; Webster and Weissberg 2001).

Both the mode of introduction and the stream flow velocity influenced the distribution of chemical exposure throughout the stream. Furthermore, these two factors were shown to interplay to create unique exposure distributions (Lahman and Moore 2015; Steele et al. 2018; Finelli et al. 2000). Chemicals appeared less likely to be distributed upward from the boundary layer under groundwater introduction treatments, particularly at low flow velocities (Fig. 6). However, such exchange appeared to be facilitated by increases in flow speed. Under high flow velocity treatments, chemicals introduced via groundwater were more likely to leave the boundary layer (Fig. 7). Higher levels of exposure were more commonly observed in the upper layers of the stream following a groundwater introduction under high flow velocity treatments. Conversely, chemicals introduced via runoff were seen to distribute throughout the water column under all stream flow velocities (Fig. 8; Fig. 9). These trends are indicative of the distribution of chemicals by turbulent flow (Vogel 1994; Sanford 1997; Moore and Crimaldi 2004). The more laminar flow within the boundary layer appeared to lend itself to lesser degrees of chemical exchange between layers of the water column (Vogel 1994). In comparison, chemicals introduced via runoff are more likely to encounter turbulence immediately at free stream velocity (Nikora 2010). Increased stream flow velocity also gives rise to an increase in turbulence and appeared to facilitate the distribution of chemicals throughout the water column (Moore et al. 1994; Vogel 1994; Finelli et al. 2000).

Variation Across Measures of Exposure

While general trends in exposure distributions emerged with shifts in mode of chemical introduction and stream flow velocity, these distributions varied depending on how exposure was defined (Harrigan and Moore 2017). Across all treatments, different definitions of exposure resulted in unique distributions of identified exposure levels throughout the stream.

The most dramatic differences were those exhibited between models created using peak area and magnitude, as compared to those created using peak frequency (e.g., left panel Fig. 6). Areas nearer to the point of chemical introduction were identified as being of the highest concern using both peak area and peak magnitude definitions of exposure. Conversely, when exposure was defined as the peak frequency, the areas identified as experiencing the highest levels of exposure were consistently observed further downstream (e.g., left panel Fig. 6). Furthermore, peak frequency was generally observed to display a more even distribution across quartiles of exposure within a given layer. These distributions often manifested as gradual gradients, radiating outward from hot spots of exposure within the stream (e.g., right panel Fig. 9). By comparison, peak area and peak magnitude tended to display a pronounced absence or prevalence of high levels of exposure in any given layer. Often distributions using these latter definitions were observed to include sharp demarcations between high and low levels of exposure (e.g., right panel Fig. 9).

While the distributions of peak area and peak magnitude were generally more similar to one another than to the distribution of peak frequency, there were still notable differences between their distributions. The demarcations between high and low concentrations were generally sharpest when exposure was defined as the peak magnitude (e.g., Fig. 9). By comparison, gradual exposure gradients were more commonly observed under distributions of peak area, although peak area exhibited more pronounced demarcations than peak frequency (e.g., Fig. 8). Additionally, distributions of peak magnitude identified areas outside of the layer of chemical introduction as experiencing high levels of exposure more commonly than the peak area (Fig. 6; Fig. 7) (Lahman and Moore 2015; Bhimireddy and Bhaganagar 2018). Peak areas falling into the uppermost quartile were only observed outside the layer of chemical introduction

under runoff introduction treatments at low stream flow velocities (Fig. 8) (Moore et al. 1994; Finelli et al. 2000). For all other treatments, these relatively high peak areas were confined to either the layer nearest the water's surface or the layer nearest the substrate, for groundwater and runoff treatments respectively (Webster and Weissberg 2001). By comparison, peak magnitudes falling into the uppermost quartile were observed outside the layer of introduction under all treatments (Fig. 6 - 8).

Overall, models created using peak area and maximum peak magnitude to define exposure were most like each other, with distributions of the peak frequency being the most unique. Defining exposure as the peak area appeared to create distributions that shared some similarities with distributions of both peak magnitude and peak frequency (Murlis et al. 2000; Justus et al. 2002). This aligns with that both the peak frequency and the size of those peaks contribute to the total area of peaks observed in a given location (Edwards and Moore 2014). Most significantly, no two measures of exposure produced the same distributions of exposure. In all observed treatments, the areas of the stream that were identified as experiencing the highest degrees of exposure varied as a consequence of the exposure definition used to construct the model (Harrigan and Moore 2017).

Implications for Ecotoxicology

Previous studies have demonstrated that variation in flow can significantly alter the structure of chemical pulses throughout stream environments (Mylne 1993; Moore et al. 1994; Gordon et al. 2012). Stream characteristics such as substrate, habitat type, and the presence of obstructions have been shown to alter the distribution of turbulence throughout the stream environment, consequently altering the patterning of chemical plumes (Denny 1993; Yee and Biltoft 2004). Furthermore, the dynamic nature of exposure has repeatedly been demonstrated to

induce observable consequences for exposed organisms (Milne et al. 2000; Ludington and Moore 2017; Neal and Moore 2017). Building on the understanding that toxicant exposure is shaped by flow, the results of this study illustrate that point of chemical entry and stream flow velocity also contribute to toxicant pulse characteristics (Sanford 1997). The incorporation of these system-specific factors will allow for more effective focusing of risk management efforts (Glasgow et al. 2004; Ashauer et al. 2007; Gordon et al. 2012). For instance, groundwater contamination was observed to remain primarily within the boundary layer at low stream flow velocities but was seen to distribute upwards in the water column with increased flow velocity (Vogel 1994). As such, in studying toxicants that are most likely to enter a stream via groundwater contamination, benthic organisms may be of particular concern in slow-moving streams (Roy and Bickerton 2011). On the other hand, in streams with higher flow velocities, organisms throughout different levels in the water column are more likely to be exposed, calling for a more generalized monitoring (Mackay et al. 1985). As flow velocity and mode of toxicant introduction give rise to variation in the structure of chemical plumes at scales relevant to many aquatic organisms of interest, the incorporation of these factors is necessary for the optimal modeling of toxicant transport and corresponding ecological consequences.

In addition to where toxicant pulses occur within a stream, the fluctuations of those concentrations at a given location over time are vital to describing exposure (Watanabe et al. 2018; Weissburg et al. 2002; Parsons and Surgeoner 1991). Previous investigations of intermittent exposures have demonstrated that temporal characteristics of toxicant pulses are critical factors in predicting a variety of physiological endpoints for a range of aquatic taxa. The duration and frequency of ammonia exposure pulses have been shown to influence the severity of morphological consequences for exposed rainbow trout (Milne et al. 2000). Green frog

tadpoles exposed to carbaryl were shown to develop and survive at different rates, depending on whether exposure was administered as a single, acute dose or multiple doses (Boone and Bridges 2003). Similarly, equivalent doses of fenvalerate, when distributed over different lengths of time, resulted in differing rates of emergence success in caddisfly larvae (Schultz and Liess 2000). Depending on the peak characteristic being described, degrees of exposure are distributed differently throughout the stream. The definition being used to model toxicant pulses throughout a stream environment can create dramatic differences in which portions of the stream are identified as undergoing the highest degrees of exposure (Edwards and Moore 2014; Harrigan and Moore 2017). As previously noted, defining exposure as either the peak area or the peak magnitude resulted in much steeper gradients between high and low levels of exposure than when exposure was defined as the peak frequency. Therefore, if an organism is sensitive to the maximum concentration of toxicant peaks, that organism will experience dramatically different levels of toxicity than an organism just a few centimeters away (Neal and Moore 2017). By comparison, if the consequences of exposure primarily depend on the number of pulses, organisms nearer to one another are likely to experience more similar levels of exposure (Boone and Bridges 2003; Hoang et al. 2007). Regardless of the mode of introduction or flow regime, different definitions of exposure resulted in different distributions of exposure levels throughout the stream (Edwards and Moore 2014; Harrigan and Moore 2017). As such, consideration of which aspects of toxicant fluctuation organisms of interest respond to warrants consideration in exposure modeling (Diamond et al. 2005; Gordon et al. 2012; Ashauer and Escher 2010).

Finally, these models can be utilized as a tool in determining when the dynamic nature of exposure in flow should be accounted for in exposure modeling. Just as the direct monitoring of all aquatic ecosystems of concern is an unpragmatic approach to resource management, infinitely

precise modeling is also infeasible. The additional dimensions of complexity added by consideration of fluctuations in exposure make dynamic exposure modeling a relatively high effort approach, compared to more traditional paradigms. As previously discussed, temporal variation in exposure is a necessary parameter in predicting exposure outcomes in some systems. Conversely, for many systems of concern, static exposure paradigms may be the most appropriate and efficient means of modeling exposure. Understanding when dynamic exposure should be incorporated into predictions of environmental toxicity is key to optimal resource management. The models constructed in this study provide a first step in identifying ecosystem factors that should inform design decisions in exposure modeling.

Conclusions

Toxicant exposure in aquatic ecosystems is highly variable in both space and time (Sophocleous 2002; Thorp et al. 2006; Schetterer et al. 2010). While many approaches to exposure modeling utilize static toxicant concentrations, exposure in nature often manifests as series of toxicant pulses (USEPA 2000; Butcher et al. 2006; Gordon et al. 2012). In flowing environments, the movement and distribution of chemicals are determined by the degree of turbulent flow interacting with the mode of entry of those chemicals (Atema 1988; Westerberg 1991; Lahman and Moore 2015). Variation in the degree of turbulent flow alters the structural patchiness of toxicant plumes within a stream environment (Sanford 1997; Yee and Biltoft 2004). This patchiness translates into variability in exposure pulses for organisms encountering the plume (Wolf et al. 2009). Across locations within a natural stream environment, the processes that give rise to chemical plume structure vary as a function of local flow characteristics (Edwards and Moore 2014).

This study demonstrates that the distribution of exposure throughout a stream environment can be greatly influenced both by how the toxicant enters the stream and the flow velocity of the stream (Finelli et al. 2000; Wolf et al. 2004; Steele et al. 2018). In addition to factors that affect the structure of the toxicant plume itself, exposure distributions can vary dramatically as a consequence of how exposure is defined. Conceptualizing dynamic exposure as a series of toxicant pulses, a variety of peak characteristics can be used to describe degrees of exposure (Zhao and Newman 2006; Ashauer and Escher 2010). These peak characteristics will be influenced differently by changes in turbulent flow, thereby creating unique exposure distributions for each peak characteristic (Edwards and Moore 2014; Ludington and Moore 2017; Harrigan and Moore 2017). Depending on which aspect of toxicant peaks most influences outcomes for exposure organisms, effective ecotoxicological modeling requires appropriately defining exposure (Harrigan and Moore 2017).

Direct measurements of fluctuations in chemical concentration over time can be used to construct three-dimensional toxicant exposure models (Moore et al. 1989; Weissburg et al. 2002; Harrigan and Moore 2017). These models can be utilized to compare patterning of exposure in streams of different flow velocities, taking both mode of toxicant introduction and organism position in the water column (benthic, mid-water column, or surface) into consideration (Lahman and Moore 2015; Nikora 2010). Models constructed under differing flow conditions reflect that, at the scale of aquatic macroinvertebrates, variation in stream hydrodynamics creates variability in the peak area, peak magnitude, and peak frequency comprising an exposure event (Wolf et al. 2009; Reidenbach and Koehl 2011; Ludington and Moore 2017). Utilizing point measurements to characterize dynamic exposure throughout artificial streams more accurately is a first step

towards understanding how flow variation influences exposure events at the scale of organisms of interest in natural environments.

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APPENDIX A - TABLES

Table 1. Percent area of each layer falling into a given quartile of observed values for each definition of exposure. Values as observed under a groundwater introduction treatment at a low stream flow velocity.

	Distance Above Substrate	Percent Area in First Quartile	Percent Area in Second Quartile	Percent Area in Third Quartile	Percent Area in Fourth Quartile
Peak Area	3 cm	35.79	10.87	3.03	50.31
	14 cm	85.43	14.21	0.35	0
	25 cm	99.89	0.11	0	0
Peak Magnitude	3 cm	7.20	38.15	0.31	54.34
	14 cm	1.45	98.55	0	0
	25 cm	5.55	2.22	29.77	62.46
Peak Frequency	3 cm	12.87	18.66	28.31	40.15
	14 cm	78.15	14.75	6.32	0.79
	25 cm	90.07	9.59	0.34	0

Table 2. Percent area of each layer falling into a given quartile of observed values for each definition of exposure. Values as observed under a groundwater introduction treatment at a high stream flow velocity.

	Distance Above Substrate	Percent Area in First Quartile	Percent Area in Second Quartile	Percent Area in Third Quartile	Percent Area in Fourth Quartile
Peak Area	3 cm	5.00	3.47	16.76	74.77
	14 cm	8.06	33.09	58.85	0
	25 cm	0.79	45.60	53.62	0
Peak Magnitude	3 cm	0	1.38	6.59	92.03
	14 cm	0	0	13.02	86.98
	25 cm	5.04	8.41	36.38	50.16
Peak Frequency	3 cm	31.73	56.36	11.91	0
	14 cm	53.56	43.18	3.21	0.05
	25 cm	72.32	12.45	10.26	4.97

Table 3. Percent area of each layer falling into a given quartile of observed values for each definition of exposure. Values as observed under a runoff introduction treatment at a low stream flow velocity

	Distance Above Substrate	Percent Area in First Quartile	Percent Area in Second Quartile	Percent Area in Third Quartile	Percent Area in Fourth Quartile
Peak Area	3 cm	0.72	12.14	10.43	76.72
	14 cm	0.97	57.07	41.96	0
	25 cm	5.55	34.59	7.76	52.10
Peak Magnitude	3 cm	4.21	0	0	95.79
	14 cm	0	8.98	2.14	88.88
	25 cm	14.19	0	0.07	85.74
Peak Frequency	3 cm	64.91	35.09	0	0
	14 cm	62.42	15.20	16.77	5.61
	25 cm	47.26	16.50	27.61	8.64

Table 4. Percent area of each layer falling into a given quartile of observed values for each definition of exposure. Values as observed under a runoff introduction treatment at a high stream flow velocity.

	Distance Above Substrate	Percent Area in First Quartile	Percent Area in Second Quartile	Percent Area in Third Quartile	Percent Area in Fourth Quartile
Peak Area	3 cm	0.02	99.98	0	0
	14 cm	44.62	55.38	0	0
	25 cm	10.38	34.45	2.19	52.98
Peak Magnitude	3 cm	0	20.58	20.19	59.23
	14 cm	13.65	76.96	9.37	0.01
	25 cm	4.73	18.82	4.92	71.54
Peak Frequency	3 cm	44.59	29.31	14.13	11.96
	14 cm	48.41	21.40	13.83	16.37
	25 cm	57.93	19.09	14.20	8.78

Table 5. Root Mean Square (RMS) of each layer for all treatments and definitions of exposure.

	Mode of Introduction	Stream Flow Velocity	Distance above Substrate	Layer RMS
Peak Area	Groundwater	Low	3 cm	704.46
	Groundwater	Low	14 cm	6.63
	Groundwater	Low	25 cm	2.05
Peak Area	Groundwater	High	3 cm	783.63
	Groundwater	High	14 cm	83.98
	Groundwater	High	25 cm	32.96
Peak Area	Runoff	Low	3 cm	335.35
	Runoff	Low	14 cm	49.53
	Runoff	Low	25 cm	703.18
Peak Area	Runoff	High	3 cm	10.42
	Runoff	High	14 cm	5.92
	Runoff	High	25 cm	703.40
Peak Magnitude	Groundwater	Low	3 cm	82010.92
	Groundwater	Low	14 cm	0.28
	Groundwater	Low	25 cm	4.37
Peak Magnitude	Groundwater	High	3 cm	81600.67
	Groundwater	High	14 cm	22007.42
	Groundwater	High	25 cm	918.24
	Runoff	Low	3 cm	1479.64

Peak Magnitude	Runoff	Low	14 cm	19336.93
	Runoff	Low	25 cm	81600.75
Peak Magnitude	Runoff	High	3 cm	558.63
	Runoff	High	14 cm	0.29
	Runoff	High	25 cm	82018.07
Peak Frequency	Groundwater	Low	3 cm	4.25
	Groundwater	Low	14 cm	1.76
	Groundwater	Low	25 cm	2.45
Peak Frequency	Groundwater	High	3 cm	2.51
	Groundwater	High	14 cm	2.87
	Groundwater	High	25 cm	1.84
Peak Frequency	Runoff	Low	3 cm	1.79
	Runoff	Low	14 cm	2.61
	Runoff	Low	25 cm	2.31
Peak Frequency	Runoff	High	3 cm	3.02
	Runoff	High	14 cm	5.02
	Runoff	High	25 cm	3.25

Table 6. Range and partial still of each layer for all treatments and definitions of exposure.

	Mode of Introduction	Stream Flow Velocity	Distance above Substrate	Layer Range	Layer Partial Sill
Peak Area	Groundwater	Low	3 cm	10.07	8.72
	Groundwater	Low	14 cm	11.07	35.62
	Groundwater	Low	25 cm	16.50	0
Peak Area	Groundwater	High	3 cm	10.00	82878.34
	Groundwater	High	14 cm	26.87	0
	Groundwater	High	25 cm	10.99	0
Peak Area	Runoff	Low	3 cm	29.94	0
	Runoff	Low	14 cm	14.12	0
	Runoff	Low	25 cm	10.00	543.45
Peak Area	Runoff	High	3 cm	36.19	62.09
	Runoff	High	14 cm	15.21	49.63
	Runoff	High	25 cm	10.00	69.84
Peak Magnitude	Groundwater	Low	3 cm	10.14	0.02
	Groundwater	Low	14 cm	52.16	0
	Groundwater	Low	25 cm	56.57	0
Peak Magnitude	Groundwater	High	3 cm	19.15	0
	Groundwater	High	14 cm	42.28	0
	Groundwater	High	25 cm	10.00	459501.54

Peak Magnitude	Runoff	Low	3 cm	44.04	0
	Runoff	Low	14 cm	44.04	0
	Runoff	Low	25 cm	19.15	0
Peak Magnitude	Runoff	High	3 cm	52.16	0
	Runoff	High	14 cm	16.50	0.10
	Runoff	High	25 cm	10.14	28276.65
Peak Frequency	Groundwater	Low	3 cm	20.63	4.85
	Groundwater	Low	14 cm	13.84	2.69
	Groundwater	Low	25 cm	15.11	7.00
Peak Frequency	Groundwater	High	3 cm	24.94	0
	Groundwater	High	14 cm	32.92	7.22
	Groundwater	High	25 cm	15.01	6.34
Peak Frequency	Runoff	Low	3 cm	29.94	0.32
	Runoff	Low	14 cm	23.31	7.12
	Runoff	Low	25 cm	54.32	9.72
Peak Frequency	Runoff	High	3 cm	13.65	8.58
	Runoff	High	14 cm	12.59	27.48
	Runoff	High	25 cm	19.15	12.62

APPENDIX B – FIGURES

Figure 1. Diagram of flume. As seen from above (far left), with focus on mixing and working sections (upper right), and with focus on the outflow (lower right). Depiction of flume as seen from above adapted from Harrigan and Moore 2017.

Figure 2. Series of adjustable nozzles used to deliver tracer solution at a constant rate of 60 ml/minute (bottom). For groundwater introduction treatments, the series of nozzles was buried approximately 2 cm deep in the substrate. For runoff introduction treatments, nozzles were suspended 10 cm above the water's surface.

Figure 3. Diagram of sampling grid as seen from above. Line intersections indicate the 20 sampling sites (4 x 5 grid) sampled within each layer in the water column. Centroid grid was aligned with the center of the flume. Samples were taken in the center of the stream, 5 cm outward from the center on both sides, and 15 cm outwards from the center on both sides. This sampling pattern was repeated 30, 40, 50, and 60 cm downstream from the delivery system. Sampling layers were positioned 3 cm, 14 cm, and 25 cm above the substrate. Substrate is represented by brown, solid layer. Select sampling points have been labeled along each dimension for illustration.

Figure 4. Example of dopamine concentration (μM) over time (s). Graphical representation of frequency (number of peaks during the sampling interval) and magnitude (highest concentration of the tracer molecule recorded during the peak). An approximation of total exposure at a given sampling site was also obtained by integrating and summing the area under each of the peaks.

Figure adapted from Harrigan and Moore 2017.

Figure 5. Map key. Blue Xs indicate delivery system location for groundwater and runoff treatments respectively. Arrow indicates direction of flow.

Figure 6. Models of exposure (defined as peak area, peak magnitude, and peak frequency) as observed under a groundwater introduction treatment at a low stream flow velocity. Models are scaled to the maximum and minimum values observed for each peak characteristic across all treatments.

Figure 7. Models of exposure (defined as peak area, peak magnitude, and peak frequency) as observed under a groundwater introduction treatment at a high stream flow velocity. Models are scaled to the maximum and minimum values observed for each peak characteristic across all treatments.

Figure 8. Models of exposure (defined as peak area, peak magnitude, and peak frequency) as observed under a runoff introduction treatment at a low stream flow velocity. Models are scaled to the maximum and minimum values observed for each peak characteristic across all treatments.

Figure 9. Models of exposure (defined as peak area, peak magnitude, and peak frequency) as observed under a runoff introduction treatment at a high stream flow velocity. Models are scaled to the maximum and minimum values observed for each peak characteristic across all treatments.