Title: A Chemical and Ecological Assessment into Elemental Loading from Ford Crossings in Ashdown Forest, Sussex, United Kingdom

Keywords: freshwater quality, biomonitoring, vehicular wash-off, Road run-off, metal loading, ford crossings

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Abstract: Numerous studies have identified the issue of road surface runoff as a source of contamination into waterways but the impact of vehicular wash-off is less well understood. A ford crossing provides a pathway for vehicle-derived contaminants emanating from both road surface runoff and vehicular wash-off into a river system. Twyford Lane Ford (Ford 1) and Birchgrove Lane Ford (Ford 2), located ca. 600m apart on a tributary of the River Ouse in Sussex (UK), were the focus of this study. A combination of biomonitoring (assessment of benthic macroinvertebrates) and chemical assessments of water and sediments has been undertaken to determine any detrimental impacts, such as a lack in biodiversity, resulting from the ford crossings. Sediment concentrations of chromium (Cr³⁺), lead (Pb) and zinc (Zn) were generally elevated at Ford 1, attenuating at sampling points between the fords to then peak at Ford 2. However, soil organic matter (SOM) and sediment particle size were seen to have an influence on elemental concentrations, in general with an increase in elemental concentrations associated with a higher percentage of fine-grained sediments ($\leq 63 \mu\text{m}$). Elevated concentrations of Zn and magnesium (Mg) were identified within water samples taken during a precipitation event following a prolonged dry period. The biomonitoring results found reduced BMWP (Biological Monitoring Working Party) scores at positions close to the ford crossings, and where the stream was in proximity to the roadside. Sensitive Ephemeroptera were largely absent at sampling points closest to the fords, which is likely to be associated with elevated Zn. The results suggest that careful consideration should be applied when selecting crossing points over sensitive waters.

Stream

Ford 1

Ford 2

Drainage Duct

100 m

1 Introduction

Water resources are coming under increasing pressure from human development, particularly in heavily populated areas such as South-East England. Roadways provide a linear source of contamination, both into the atmosphere and onto the neighbouring land and waterways. The UK Department for Transport (2016) estimated the total road length in Great Britain to be 246,000 miles with 2214 ford river crossings. Such fords provide a convergence point for traffic-derived contaminants to enter waterways via vehicular wash off, road surface runoff or from airborne particulate matter. Sebastiao et.al. (2017) identified first flush run-off events, i.e. a rainfall event preceded by a prolonged dry period, as a major contribution to metal loading into waterways. Vehicle-derived trace metals and metalloids delivered into the environment emanate from a variety of sources: for example, dissolved arsenic (As), cadmium (Cd), copper (Cu), Pb, nickel (Ni) and Zn from oils and lubricants and Cu from brake pad linings and tyres (Johnsson et al., 2002; Davis et al., 2001). Galvanised metals on vehicles provide a source of Zn and Cd (Beasley & Kneale, 2003). Additionally, applying weed killers on roadsides can provide sources of As, Cd and Cu (Beasley & Kneale, 2003).

A study by Revitta et al. (2014) identified land use type as the main contributing factor to metal loading in river systems. Urbanised areas were associated with increased levels of metals, predominantly derived from vehicle and road surface wear. Vehicles are sources of particulate matter pollution, often deposited on road surfaces to be subsequently washed into neighbouring drainage systems or local waterways (Adamiec et al., 2016). The main contaminants of concern associated with vehicles are heavy metals and fuels. The Department of Environment (DoE) industry profiles (CL:AIRE 2018) do not specifically cover roadways, but profiles for garages and filling stations, and transport and haulage centres,

highlight vehicle-derived contaminants. Table S1 in the supplementary information (SI) highlights the contaminants that are likely to find their way into water systems in proximity to roadways. Metals are of particular concern and are becoming more prevalent due to their widespread use and accumulative behaviours (Kang et al., 2017). Heavy metals have the propensity to adsorb to sediments, which thus provide a sink or, through the process of desorption, a subsequent source of metal contamination in waters.

The loading of contaminants into an ecosystem is an arresting factor for ecological succession which interrupts biological resources, including agricultural and fishery products (Amiard-Triquet et al., 2011). Identifying an abundance of tolerant species gives insights into sustainability of the biodiversity and indicates ecosystem performance, potentially giving warning, and thus enabling some form of mitigation. A pollution event into a stream, e.g. an influx of contaminants emanating from a point source, can, and often does, create a dead zone within the river system which effectively creates fragmented habitat zones. This stops migratory paths and, particularly for species that live the entirety of their lives in the river, arrests their dispersal and colonisation opportunities. Debenski and Holt (2000) identified that species which are displaced due to detrimental anthropogenic pressures tend to accumulate in the remaining patches of “better quality” habitat. However, after an initial abundance peak, it was found that population size soon regressed to the mean (termed a relaxation period, Primack, 2006) due to further dispersal limitations, inter/intraspecies relationships, genetic pressure and increased vulnerability to catastrophic events. Riparian ecosystems, one of the most diverse zones in the UK, interact with the landscape around them and often provide the most critical nutrient and trophic interactions in an ecosystem (Naiman et al., 2005). Therefore, their preservation should be sought with priority.

Lauma & Rainbow (2010) argue that trace metal contamination is poorly understood, attributing this to the lack of application of toxicity data to the complexities of natural water systems and biological communities. Wright et al. (1997) state that biological monitoring is required to complement chemical monitoring, as a biotic appraisal gives a means to determine the consequences of environmental stress. Therefore, a combination of both methods should give comprehensive insight to the chemical regime of a water body, helping to elucidate the governing factors. Because of the ever-increasing reliance on vehicles, and consequent environmental pressures, this study aims to establish the impact of vehicular wash-off and surface runoff emanating from ford river crossings on the elemental and macroinvertebrate composition of the adjacent stream, located within the Birchgrove Area of the Ashdown Forest, East Sussex (Figure 1). The study integrates geochemistry, fluvial dynamics, geomorphology and ecology - thus providing a holistic appreciation of metal contamination often missing from many works, in addition to providing rare insight into the effects that ford crossings have on the local water environment.

2. Description of the study site

The site occupies a section of a tributary of the River Ouse, located in a shallow valley - comprising campestrial, sylvan and agrarian land - near Twyford Farm in the Ashdown Forest, Sussex (National Grid Ref. for central point: 539940, 130620). Local topography slopes gently down from the north-east to the south-west with elevation ranging from 70 to 100 m above sea level.

The north-eastern section of the study site (Ford 1) is located within an area of deciduous woodland, which turns into coniferous silviculture to the south-east (Figure 1). As the

stream courses toward the south-west, it is flanked on both sides by sheep pasture, part of Twyford Farm. Hydrophilic lower order plants and trees occupy the banks of the stream as it traverses the grass pasture. Flowing south-west past Birchgrove Lane (Ford 2) the stream courses into privately-owned land forming part of Hurstwood Farm. This area comprises unkempt grass and ruderals with several broadleaf standards. Land to the south-east of Hurstwood Farm supports deciduous woodland.

Figure 1 shows the site location, and sample stations. Figure S1 in the SI shows site topography. The main channel is joined immediately upstream of Ford 1 by a minor channel (Minor Stream 1 [MS1]) which flows south-east to form a confluence at Ford 1. MS1 is cut adjacent to Twyford Lane to assist with road drainage. Minor Stream 2 (MS2) joins the main channel immediately upstream of Ford 2. MS2 feeds from a fishing lake located south of Grinstead Wood. Historic and contemporary mapping identifies a ford crossing on MS2, however, the site walkover identified this as a culvert running under a track to which vehicular access was not permitted. Minor Stream 3 (MS3) is another cutting into the roadside, that runs alongside Birchgrove Lane before entering MDPE ducting and then draining downstream of Ford 2. Both MS1 and MS3 are ephemeral and were dry on several visits.

With reference to online mapping published by the British Geological Society, the geology underlying the site comprises interbedded sandstones and siltstones of the Ashdown Formation, part of the Lower Cretaceous Wealden Group . However, superficial alluvium occupies the streambed and banks. It is assumed that groundwater flows southward, following the course of the river. Intermittent clay horizons may also give rise to perched groundwater. Likely coefficients of permeability, with reference to Knappett and Craig

(2012), are 10^{-5} to 10^{-8} m/s for alluvial deposits, 10^{-7} to 10^{-9} m/s for the clays and silts of the Ashdown Beds and 10^{-2} to 10^{-6} m/s for the sandy horizons.

Twyford Lane and Birch Grove Road are unclassified minor roads that cross the stream via two fords and thus represent potential sources of contamination of stream water. Traffic counts were conducted at hourly intervals on various days and times (Table S2 - SI) to obtain an estimate of traffic density - an average of 144 crossings per day at Twyford Lane (Ford 1) and 223 at Birchgrove Lane (Ford 2).

Several historic discharge consents, mostly for sewage/effluent, have been identified (Table S3 - SI) within or upstream of the site. Said discharges may lead to the release of adsorbed metals, or to an increase of their adsorption. Additionally, the influx of nutrients may lead to reduced dissolved oxygen which would impact stream ecology. However, since the identified discharge consents date back to 2006 and 1961, the stream system has had ample time to recover.

A conceptual site model (Figure 2) identifies several source-pathway-receptor (SPR) linkages. Humans and their pets are recognised as receptors through direct contact with waters and/or sediments potentially impacted by heavy metals. Several site visits identified humans and dogs playing and bathing in the waters of the fords. Nevertheless, human risk is considered to be low. On the other hand, there exists significant potential risk to ecology, both faunal and floral.

3. Methods and Materials

3.1 Soil and water sampling

17 samples of each, for water and near surface sediments, were collected (Figure 1), at locations recorded using handheld GPS in reference to WGS84 datum (Tables S4 and S5 - SI).

Selection of sample sites reflected distance from the ford crossings, starting with a control sample at sample point 1 (SP1). Water and sediment samples were taken immediately adjacent to the upstream and downstream section of the fords and at intervals downstream. Other sampling positions were selected to determine attenuation characteristics; SP17 was specifically selected to delineate any influence from the historic discharge consent. Finally, five near surface soil samples (between 0.20 – 0.50 m bgl) were taken within the surrounding area to constrain local background metal concentrations.

Water samples were collected on two separate occasions, during a prolonged dry period (12/05/18) and during a precipitation event (30/07/18). Because the stream is likely spring fed, it is likely unlikely that it dries out – further backed up by sampling taking place during the summer. Water samples were collected in a 1 L amber glass bottle and held in temperature-controlled storage (cool box with ice packs). Dissolved oxygen (DO) was measured on site using a CHEMets DO kit and Electrical Conductivity was measured using a portable Hanna EC meter. Stream discharge measurements were estimated using a float over a 10 m reach. Metal concentrations were subjected to two tailed t-tests to identify the significance of first flush runoff.

Sediment samples were collected, from downstream to upstream to avoid cross contamination, using a plastic trowel cleaned prior to each location. Three samples of sediment were taken at the base of each bank and at the bed under the thalweg, these were combined to form a composite sample, from no deeper than 50 mm depth - the optimum oxidising and most biologically active zone (Naiman et al., 2005). Once collected, sediment was placed in a 500 ml amber jar and stored in a cool box. Additionally,

approximately 1 kg of sediment was placed in a plastic soil bag for determination of particle size distribution (PSD).

Distances of sampling locations from the nearest road and ford crossing were measured using a GPS and ArcGIS. The results were used to calculate Pearson's correlation coefficients to assess the influence that roads and ford crossings had on metal concentrations.

3.2 Laboratory Analysis

Samples were sent to UKAS accredited laboratory - The Environmental Laboratory (E-lab), St. Leonards-on-Sea, East Sussex - for chemical testing. The analysis suite comprised zootoxic and phytotoxic elements (As, Cd, Cr, Cu, mercury (Hg), magnesium (Mg), manganese (Mn), molybdenum (Mo), Pb and Zn) associated with vehicular runoff/wash-off. Extractable/dissolved metals were determined using ICP-MS whilst hexavalent Cr was measured using colorimetry. The methodology statements (including estimates of accuracy and precision) from E-lab are presented in the SI.

Sediment particle size distribution (PSD) tests were carried out at a UKAS accredited geotechnical laboratory – K4 Soils Laboratory, Watford, Hertfordshire – in accordance with British Standard 1377-2 (1990). Samples were passed through a series of sieves ranging from 125 mm down to 63 µm, the size ranges were then quantified by percentage of total weight.

3.3 Ecological Sampling

Kick sampling was selected as it is the most widespread and well-developed methodology (Mason, 2002), is often applied in order to comply with policy-driven guidelines (Carter and Resh, 2011) and has the lowest resource requirement. Sampling locations were identified using GPS, once identified one person (the kicker) stands 3m upstream of the other (the

netter). The kicker proceeds to manoeuvre in a zigzag pattern kicking up the sediment, causing as much disturbance as possible (the zigzag pattern helps ensure that the different habitats are disturbed) whilst the second person nets the dislodged invertebrates flowing down the stream. The net had a 250mm x 250mm frame with a 900µm mesh net (275mm deep) and a 1.60m handle. The invertebrates were then placed into a sampling tray and taken back to the van (serving as a mini-lab) for identification and abundance. An electronic microscope with 1000x magnification was used to aid with identification and to take photographs. All aquatic invertebrates were returned to their relative locations once identification had been completed. The presence and abundance of species were then used to assign a BWMP score and to determine the relative abundance of certain species.

BMWP scores for taxa can be found in Henderson, 2006 pp.127. With reference to the Ouse and Adur Rivers Trust, the classifications for BMWP scores are >130 – Very Good, 81 – 130 – Good, 51 – 80 Fair, 11-50 – Poor, and 0 – 10 Very Poor.

4.0 Results

4.1 Chemical Composition of Water

As, Mo, Cr, Pb, Cu and Hg along with hexavalent Cr were uniformly below their respective detection limits (5 µg/L for As, Mo, Cr and Cu, 1 µg/L for Pb, 0.1 µg/L for Hg and 100 µg/L for hexavalent Cr).

Zn levels in water were generally below the detection limit (5 µg/L) (Figure 3). However, peak concentrations, particularly after the precipitation event, were shown at sampling locations immediately downstream of the fords, SP4 and SP11 (22 µg/L and 38 µg/L respectively). Zn concentrations returned to levels below the detection limit with increased distance downstream from both fords.

Mg levels generally declined from SP1 to SP14 in dry conditions along the main stream (Figure 3), suggesting a source upstream from SP1. No outliers were observed in dry conditions. A significant increase was identified after the precipitation event. In wet conditions, Mg spiked at SP4 and SP11 showing concentrations of 4250 µg/L and 4510µg/L respectively – these were the highest recorded concentrations of Mg in water. The Mg concentrations for SP4 and SP11 were 3150 µg/L and 2960 µg/L respectively in dry conditions and were not significantly different from other points. SP3 had a high concentration of Mg in wet conditions, possibly due to the proximity of Twyford Lane. Mg concentrations generally decreased with distance from the fords.

Mn results ranged between <5 and 998 µg/L during dry conditions and <5 to 1010 µg/L in wet conditions (Figure 3). SP3 was an outlier in both, and recorded concentrations of 998 and 1010 µg/L respectively. Mn concentrations were slightly elevated after the precipitation event at SP4 and SP11, at 52 and 45 µg/L respectively when compared to 6 µg/kg at both positions during a dry period. Mn levels were relatively high at SP14 during both wet and dry conditions at 28 µg/L and 32 µg/L respectively.

4.2 Sediment Chemistry

Five control samples were taken from the underlying Ashdown Formation, the sole geology up- and down-stream of the study sites, at depths of between 0.30 m to 0.60 m and were tested for As, Cr, Cu, Pb, Mg, Mn, Ni and Zn. The results are summarised and compared to the ford river crossing results (this included all results from 1 m upstream and 10 m downstream of each respective ford) in Figure 4. Without exception, it was found that

elemental concentrations in sediments were above the average background level from sediments near the ford crossings. However, all elemental concentrations identified in the sediments were below varying human health criterion and estimated background chemistry (with reference to an Envirocheck Datasheet), shown in Table 1.

The impact that the fords and roads were having on contaminant loading into the main channel is illustrated in Figure 5. Figure 5 shows an inverse association between the concentration of elements and distances of road, and also shows significant correlation between the concentration of metals and distances from the fords. Of the metals, Zn, Pb and Cr have the most significant correlation.

The mean concentration of Pb was $16 \text{ mg/kg} \pm 6.15 (1\sigma)$, reflecting moderate spatial homogeneity. No outliers were identified. Figure 5ai suggests that Pb was most closely associated with road proximity, given the R^2 value of 0.6. Figure 5aii shows a significant relationship between downstream distance and Pb concentration. The spatial distribution map (Figure 6) suggested an influx of Pb at both ford crossings, particularly Ford 2. Pb concentrations at SP9 and SP12 were the highest recorded, when compared to background levels, in the major stream at 27 mg/kg and 24 mg/kg respectively. SP11 was slightly lower at 20 mg/kg .

The mean value for Cr was $19 \text{ mg/kg} \pm 12.35$, which suggests high spatial heterogeneity. No outliers were identified. Cr concentrations displayed the second strongest inverse relationship with distance from the road with $R^2 = 0.58$ (Figure 5bi), suggesting a significant relationship. There was a similar inverse relationship between Cr distance downstream from the fords, $R^2 = 0.63$ (Figure 5bii), the strongest for the metals considered here. At Twyford Lane Ford (Ford 1) Cr concentrations of 28 mg/kg and 35 mg/kg were recorded

232 immediately upstream (SP2) and downstream (SP4) respectively. Similarly, at Birchgrove
233 Lane Ford (Ford 2) concentrations of 33 mg/kg and 37 mg/kg were recorded at SP9 and SP11
234 respectively. The maximum Cr concentration, 41 mg/kg, was recorded at SP12. Cr, in
235 general, decreased downstream with increasing distance from the crossings. Upstream of
236 both, Cr was recorded at 4.0 mg/kg (the lowest concentration) and at SP14, 100m
237 downstream from Ford 2, Cr was noted at 5.7 mg/kg. Spatial distribution maps (Figure 6)
238 identified both fords as having increased Cr concentrations relative to background levels.
239 Concentrations peaked at the crossings before they gradually decreased downstream with
240 distance of the ford. Moderate concentrations of Cr were identified in the MS2.

241 The mean concentration of Zn was 94 mg/kg $\pm$ 38, indicating high spatial heterogeneity,
242 although no outliers were identified. Figure 5ci shows an R^2 value of 0.53 indicating a
243 moderate relationship between road distance and Zn concentrations. Spatial analysis
244 (Figure 6) highlights elevated Zn, when compared to background levels, downstream of
245 both ford crossings, with a minor increase also being located at SP8. MS1 provided a large
246 influx of Zn with concentrations of 142 mg/kg and 134 mg/kg respectively at SP16 and SP3.
247 Concentrations at SP14 were higher than expected at 96 mg/kg.

248 Mn, Mg, As, Ni and Cu all had low R^2 values both for road distance and for ford distance.
249 However, all had heterogeneous, and mostly higher than background concentrations,
250 nearer to the road or ford crossing, which decreased abruptly with distance.

251 Cu, As and Ni all had no discernible relationship with ford distance, their high
252 standard deviations indicated a high spatial heterogeneity with no correlation to the fords.
253 Cu and Ni did not show evidence of loading at the fords. The highest Cu concentration (12
254 mg/kg) was recorded at SP12.

A consistent feature of the spatial distribution diagrams (Figure 6) was the influx of metals from MS1 flowing from Twyford Lane (SP16 and SP3) and joining the mainstream course immediately before Ford 1. However, both ford crossings and their road surfaces were evidently a source of Zn, Pb, Cr and, to a lesser extent, Mg to the river system.

4.3 Macroinvertebrate samples

A strong negative correlation between BMWP score and distance from the road was identified by an R^2 value of 0.56 (Figure 7a), and Figure 7b identifies a significant correlation between increasing distance from the ford crossing and increased BMWP score. BMWP results downstream of the fords were generally lower than those upstream (Figure 7c). BMWP scores of 110 and 135 were recorded 10 m and 50 m upstream of Twyford Lane Ford (Ford 1) respectively, whereas downstream of the ford, scores of 55 and 65 were recorded at the same distances. In general, with increased distance downstream from Ford 2, BMWP scores increased. An exception is the site 10 m downstream from Ford 2 which had a lower BMWP score (21) than 0.5 m and 5 m downstream (36 and 38 respectively). These lower BMWP scores correlate with increased concentrations of Cr, Pb and Zn.

Because of the limitations of BMWP scores, species/family analysis was undertaken. Figure 8 highlights presence on the basis of the following scale created by the author:

A – Abundant: The presence of such species is numerous in the sampling tray and represent a vast, often dominant, proportion of the total invertebrates' c.20+ individuals.

F – Frequent: A species which is present in fairly large quantities and easily identified within the sampling tray c.10-20 individuals.

277 **O – Occasional:** A species which is present relatively small quantities usually 3 – 9
278 individuals.

279 **R – Rare:** A species where only 1 or 2 individuals are identified.

280 **Abs – Absent:** The species was not identified at the sampling point.

281 Figure 8 shows that Ephemeroptera, is mostly rare in Ford 1 and absent in Ford 2, becoming
282 abundant with distance from the ford. Plecoptera expresses a similar frequency, however, it
283 is not recorded in abundance anywhere in the stream. Trichoptera lessens in abundance
284 immediately downstream of both fords but is occasional and frequent immediately
285 upstream in Ford 1 and Ford 2 respectively.

286 Pearson's Coefficient was used to determine the relationship between various metals and
287 BMWP score, the results are summarised in Table S11- SI. Table S11 - SI identifies that Zn,
288 Pb and Cr have Pearson's Coefficients with BMWP score of -0.77, -0.75 and -0.71
289 respectively. From this, it is determined that the concentrations of these metals have
290 significant negative impacts on BMWP score. Mn and Mg had Pearson's Coefficient scores
291 of -0.52 and -0.37 respectively, showing a quite strong negative correlation. Elements As, Cu
292 and Ni had correlations of -0.28, -0.18 and -0.02 respectively showing weak to negligible
293 negative correlations.

294 Figure 9 indicates that Cr, Pb and Zn have an influence on species composition. Mn, Mg
295 have some influence on species composition whereas As, Cu and Ni do not show discernible
296 influence.

297

4.4 Other Variables

Results of DO (Table S12 - SI and Figure S2 - SI) were consistent at all sampling points, ranging between 7 and 9 mg/l, with the exception of SP3 which had a DO level of c. 3 mg/L. Additionally, no other evidence of depleted DO levels, such as an algal growth, were observed. TDS results, converted using an electrical conductivity meter, were in the range of 158 and 203 mg/L, this was commensurate with the expected TDS level for hard water.

On average, pH was slightly higher during dry conditions (Table S13 - SI and Figure S3 - SI). The majority of results fell between the optimum range for freshwater of pH6.5 – pH7.5 (Namiesnik and Szefer, 2010).

Flow was recorded at all sampling locations on two separate occasions (after prolonged dry and wet periods) and was broadly in the range of 0.023 – 0.068 m/s at the main channel (Figure S4 -SI and S5 - SI).

Particle size distribution (PSD) and particulate organic carbon (converted to soil organic matter –SOM) tests were undertaken at each location where sediment samples were taken. No discernible relationship between fine particles, SOM and the three most prevalent metal concentrations (Zn,Pb & Cr) was obtained (Figure 10 and Table S14 - SI). The results of correlation coefficients to discern relationships between particle size, SOM and element concentrations (Table S15 – SI) highlights very weak to weak correlations between metal concentrations and soil variables in most instances. Exceptions were Cr and Mg which both had a moderate negative correlation with percentage of SOM. Additionally, As, Pb and Cu had moderate positive correlations with percentage of fine sediment particles.

5. Discussion

5.1 Comparisons with Guideline Criteria

Results from the sediment samples were compared to several criteria (Table 1): Estimated Background Value (from Landmark Dataset), SGVs and C4SL or SSAC depending on which gave the lower value. No metal concentration exceedances, deemed hazardous to human health, were identified. However, concentrations considered detrimental to human health do not necessarily pertain to macroinvertebrates. In most instances, macroinvertebrate communities are more sensitive. Cadmus et al. (2020) identified that the sensitivity of aquatic invertebrates was size dependent, therefore, it is assumed that human health guidelines are further removed for the sensitivity thresholds of aquatic invertebrates at various life cycle stages. As such, the comparisons made in Table 1 are used solely as an indicator.

The results of the chemical analysis for determinands known to be phytotoxic are summarised in Table 2, together with the respective adopted Generic Assessment Criteria (GAC) for plant growth. The compliance criteria set out in BS3882:2015 are pH dependent and thus the GAC used relate to the pH range measured on samples recovered from the site. No exceedances in determinand concentrations that are considered phytotoxic were identified.

Ranges of elemental concentrations were compared to ranges taken from other rivers around the world as identified in Pandey et al's. (2019) study Table S16 - SI. In general, the metal concentrations were lower in the present study than those of Pandey et al's study,

however, it should be noted that the present study is focused on a tributary in a rural setting rather than a main river channel that passes urban areas.

5.2 Impact of ford river crossings on heavy metals and macroinvertebrate composition

The elemental concentrations observed above background levels are likely to be attributable to the ford crossing and nearby road network, and therefore, may influence ecological composition. Road surface runoff has often been identified as a key source of metal loading into waterways (Haus et al., 2006; Stucker, et al., 2016 and Sebastiao et al., 2017). In addition, fords are generally low-elevation parts of the roadsand accumulate pollution from upgradient. It is also apparent that water quality of the channels downstream of the fords has been affected by vehicle-derived contaminants, the impact of which seems to be exacerbated by precipitation events which wash road dust into the waterways.

5.2.1 Water Chemistry

Influxes of heavy metal contamination during dry conditions were not encountered, however, wet conditions did experience influxes of Zn, Mg and Mn.

The concentrations of Zn in water were generally below detection (5 µg/L), however, in proximity to the ford crossings vehicular wash-off apparently contributes to overall Zn concentration. Of equal or more significance is the greater concentration of Zn and Mg recorded at the main channel during a precipitation event following a prolonged dry period, recorded at 22 µg/L and 38 µg/L Zn , and 4250 µg/L and 4510 µg/L Mg at sampling positions

364 located immediately downstream of Ford 1 and Ford 2 respectively. Although such
365 concentrations are below their respective threshold values, they do indicate localised
366 loading when compared to background concentrations. This pattern was not observed for
367 Mn. Sample point SP3, at Minor Stream 1 (MS1), recorded concentrations of Mn at 998 µg/L
368 and 1010 µg/L in dry and wet respectively, far higher than the next highest concentration of
369 52 µg/L at SP4. This suggests a source of Mn upstream in MS1. The desk study (Tables S17
370 and S18 -SI) did not identify any industrial activity, either contemporary or historical, near
371 MS1 other than farms. Although Mn is ubiquitous in the environment (O'Neal and Zheng,
372 2015), occurring in soil concentrations of 40 – 900 mg/kg (O'Neill, 1998), the spike in Mn
373 could have come from the use of agricultural fungicides in the adjacent agrarian land.
374 Alternatively, elevated Mn concentrations are a likely product of low DO levels (3 mg/L)
375 leading to a reducing environment allowing Mn to be liberated from the weathering of local
376 geology thus dissolve more readily. By comparison, Mn concentrations fall to 13 µg/L (dry)
377 and 26 µg/L (wet) at SP2, which had a DO of 7 mg/L and is located 5 m downstream of SP3
378 in the confluence at MS1 and the main channel. The main channel has a discharge of 0.068
379 m³/s at SP2 which dilutes the water from MS1 (discharge of 0.0032 m³/s) resulting in lower
380 Mn concentrations and higher DO levels.

381 pH levels of the water were generally within the optimum range for freshwater. The
382 expected slight fall of pH after rainfall was most pronounced around Ford 2, probably due to
383 less vegetation cover to intercept the rainfall which, with its lower pH (avg. pH 5.6), would
384 mix with river water to slightly reduce the pH. The lowest pH recorded was 6.2 (see Table
385 S13 in SI). Gerhardt (1992) identified ion regulation to be disturbed by H⁺ and Al³⁺ ions which
386 are elevated in acidified waters, a subject for further study. Further testing may identify if

prolonged rainfall acidifies the stream waters, therefore altering the chemical interactions, over a seasonal time scale.

Elemental concentrations do not exceed thresholds from the European Water Framework Directive (WFD), but do indicate (see t-test Table S19 - SI) that first flush run-off is an important contributor to metal loading into waterways, as discussed by Haus et al., (2006). Concentrations of Mg and Zn experience exponential fall toward the mean value with increased distance from each ford , another indicator of the surface run-off being the source. Paired two sample t-tests for concentrations of Mg, Mn and Zn, comparing wet and dry conditions with a 95% confidence level, suggest that precipitation events do indeed lead to influxes of Mg and Zn. The result for Mn is outside the confidence interval, likely influenced by sources from Minor Stream 1.

5.2.2 Sediment Chemistry

Numerous studies, including Sebastiao et al. (2017) identify surface runoff as a major source of metal contamination. Metal concentrations at their study site, Righters Mill Road Ford (RMRF) in Philadelphia, USA were significantly higher than those at Twyford Lane (Ford 1) and Birchgrove Road (Ford 2) despite having similar traffic volumes. However, both studies have demonstrated significant correlations between sediment metal concentrations and proximity to ford crossing and roads. Land use setting is the most likely explanation for the differences in concentrations. RMRF is located in a suburban area with a dense road network, as opposed to the rural setting of Twyford Lane and Birchgrove Road. This corresponds with the observations of Stucker et al. (2016), who identified significantly increased concentrations of Cr, Pb and Mo in waterways near to roads and found land use type to be a main driver of dissolved metal concentrations. It seems sensible that higher

concentrations of sediment metals are found in an urban setting. However, the recognition that road-derived elements are heavily influenced by particle size distribution, perhaps even more so in terms of their recalcitrance, enabling them to reside for prolonged periods within sediments, and should not be underplayed Wang et al. (2019).

Pb, Cr and Zn all show a marked increase in sediment concentration around both ford crossings, particularly at those closest to the downstream sampling point. The spatial distribution diagrams (Figure 6) identify the crossings as accumulation points for these metals. Generally, concentrations of metals increase at Ford 1, then decrease with distance downstream, only to spike again at Ford 2 subsequently regressing to the mean value, as predicted in the conceptual model (Figure 2). It therefore seems probable that the ford crossings are indeed sources of increased metal loading, particularly of Pb, Cr and Zn. Attenuation distance graphs (Figure 11) estimates that Zn returns to background range by ca. 450 m, but the critical distances for Cr and Pb are significantly shorter at ca. 106 and ca. 180 m respectively.

Concentrations of some elements recorded at SP12 (10 m downstream from Ford 2) were higher than at SP11 (immediately downstream of Ford 2 where slightly lower concentrations would be expected). For example, Pb and Cr at SP11 are 20 mg/kg and 37 mg/kg respectively, whereas at SP12, concentrations were 24 mg/kg and 41 mg/kg respectively. The concentrations then decline downstream at SP13 and SP14 (Figures 10). SP11 predominantly comprised a hard-standing bed (concrete) with a faster flow (0.046 m/s compared to 0.031 m/s at SP12) which is unfavourable for the deposition of fine-grained material (Hugget 2007). Lower metal concentrations at SP11 are therefore possibly the result of a lack of clay and SOM which would impede adsorption (Yao et al., 2015). PSD

measurements (Figure 10 and Table S14 – SI) show that SP11 had 1.3% fines while SP12, located 10m downstream of Ford 2, had 19.6% fines. Furthermore, this also corresponds with the findings of Wang et al. (2019) who found preferential accumulation of lead and zinc in finer road-deposited sediments, however, their study was undertaken to a finer resolution (<38.5 µm for Wang et al., 63 µm for this study).

Differences in SOM were not so pronounced (see Figure 10). SP14 (110 m downstream of Ford 2) has shown slight peaks in concentrations of As, Cr, Cu, Mg, Ni and Zn, which may also be related to its relatively high clay and SOM contents. However, it should be noted that organic material can serve as a source of elemental contamination through the manure from adjacent sheep pasture (Robinson et al., 2000).

Pearson's coefficients (Table S20 - SI) confirms elevated concentrations of Pb, Zn and Cr to be linked with proximity with ford crossings (i.e. concentrations decrease with increased distance). Mn and Mg were moderately associated with fords. SP11 (closest downstream of Ford 2) was identified as having increased Mg when compared to background levels (6200 mg/kg), though of no concern with regards to toxicity. SP11 is located near the drainage duct running alongside Birchgrove Road, from which sediment samples recorded relatively high Mg concentrations (2940 mg/kg), suggesting that the drainage duct is a secondary source of Mg, but it is unknown where from.

5.2.3 Pollution Sources

The elements identified at increased concentration, in comparison to background levels, have been found to be associated with vehicles in previous studies. Although Pb is no longer

found as an additive in fuels in the UK, it has been found adsorbed onto roadside soils due to its long persistence (Loganathan et al. 2013). This would act as a contributing factor to increased Pb concentrations. Sutherland and Tolosa (2001) found a linear decrease in Zn and Pb concentrations when comparing concentrations of elements with road distance, corresponding with decrease of the same elements in this study. This indicates that vehicular pollution is the likely source of two of the metals most closely associated with arresting ecological succession in this study, therefore, impeded dispersal and colonisation.

Although all elements have been associated with vehicles in previous studies, it is likely that the elements incorporated within the river sediments are derived from a variety of anthropogenic and natural sources. However, with reference to Wang et al (2019), road deposited sediments accumulate on road surfaces during dry periods and present differences in dissolvability depending on their residence sites and bonding. As a result, Wang et al (2019) found that almost half of Ni and 7% Zn found within road deposited sediments had been transported in the dissolved form via storm water. This finding correlates with the concentrations of dissolved Zn found in proximity to the crossings and also with the low concentrations of Ni found within the river sediments.

Road-derived sediments are also a factor of the roadway material. Loganathan et al. (2013) suggested that mineral aggregates, comprising different rock materials, have differing concentrations of metals. The wearing down of these aggregates have the propensity, albeit at different levels, to contribute metalliferous materials into the road dusts as asphalt is worn down. Study into the elemental composition of the road material around the two fords may identify the significance of such variability.

5.2.4 Biodiversity

478 Charlesworth et al. (2003) identified the elements with the highest potential for
479 bioavailability in their study in two urban rivers in the UK to be Cd>Zn>Cu>Pb>Ni. Their
480 study found that only a small fraction of the heavy metals was bound to readily
481 exchangeable sites. The Zn identified in Charlesworth et al's. (2003) study was found to be
482 associated with carbonates, hence its increased bioavailability. However, in the current
483 study Zn was not found to correlate with SOM (using TOC as a proxy) but was associated
484 with a decrease in biodiversity. Clearly, more work is required to identify speciation of
485 elements found to negatively impact the ecology.

486 Spot sampling and analysis of water and sediments gives only a snapshot of the metals in
487 the system. Biomonitoring allows longer-term estimation of water quality due to the
488 responses of sensitive invertebrates. Although zinc is an essential element, the WHO (2001)
489 identifies concentrations of 20 – 50 µg/l to have chronic effects on cladocerans and
490 concentrations of 50 – 100 µg/l to have chronic to acute effects, depending on water
491 hardness, on freshwater invertebrates. Additionally, several studies (Armatage, 1980;
492 Malmqvist & Hoffsten., 1999; & Beasley & Kneale., 2002) have identified Zn as the element
493 having the most pronounced negative effect on taxonomic diversity, further backed up by
494 Clements (1994) who demonstrated a distinct decline in biodiversity associated with
495 increased levels of Zn.

496 The Pearson's Coefficients (Table S21 in SI) suggest that some of the elements studied may
497 have a direct negative impact on biodiversity. Zn followed by Pb and Cr were shown to have
498 the strongest negative correlation. Hexavalent Cr (Cr⁶⁺), the more mobile, more bioavailable
499 and more toxic form, was below detection limit (0.8 mg/kg) in all samples (possibly due to
500 reduction by SOM), thus the Cr detected was trivalent. The impact of Zn in reducing the

501 BMWP score accords well with the results of Clements et al. (2013). Zn has long been
502 identified as one of the most mobile metals (Langston & Bebianno, 1998), forming inorganic
503 and organic complexes – many of which are soluble. Zn ions have been found to lead to
504 hypocalcaemia in fish by competing with Ca ions (Hogstrand et al., 1993), which could lead
505 to localised habitat fragmentation.

506 Ni had almost no impact whilst Cu and As had a negligible impact. Mn and Mg were
507 identified as having a moderate impact on biodiversity, but both have low toxicity. As such,
508 the moderate Pearson's Coefficient score associated with Mn and Mg and their respective
509 BMWP scores may be more of a product of their higher concentrations towards the ford
510 crossings and the relationships with the other metals.

511 The increased concentration of some metals and decrease in BMWP score do appear to be
512 intrinsically linked. Other variables such as discharge, EC, river bedload and shade cover do
513 not show any significant relationship. The dissolved oxygen tests did not show evidence of
514 organic pollution. Flow and bedload did not appear to have any observable impact on
515 biodiversity in the absence of elevated heavy metals. An invertebrate species that colonises
516 appropriate areas, e.g. *Philopotamidae* - a taxonomic group known to inhabit faster flowing
517 stretches - was largely absent from areas of the main channel with a low velocity, even
518 when BMWP scores were high. *Philopotamidae* however, was found in higher velocity areas
519 suggesting that colonisation had been enabled and that velocity does impact composition of
520 species but not overall biodiversity, as other species will take their place.

521 Although the BMWP score is a good indicator of habitat quality, the score does not give
522 indications of species with high tolerances and, therefore, has the potential to give
523 misleading results. As such, the identification of individual species with known tolerances is

of importance. Ephemeroptera have long been identified as a metal-sensitive taxon (Clements, 1994 and Costas et al. 2018). Figure 8 highlights a marked absence or rare occurrences of Ephemeroptera at sampling points near the fords and roads. Such absences and rarities in Ephemeroptera are correlated with areas where higher concentrations of Zn, Pb and Cr have been recorded, suggesting that these metals are, at least partly, a factor of the scarcity of Ephemeroptera. Trichoptera are considered more resistant to contaminants and, although not particularly abundant at any location, were recorded more frequently in proximity to the fords, filling the niche left from the absent Ephemeroptera. Plecoptera, observed to have variable sensitivity, were found to be absent in locations closest to the fords and to have occasional abundance in locations closest to roadsides. Conversely, Plecoptera were noted to be abundant or frequent at locations with increased distances from the fords. The biomonitoring results therefore suggest that the ford crossings are having a negative impact on biodiversity as a result of metal loading.

5.3 Implications for river ecology and environmental protection

A conceptual contamination model and interaction of various environmental components is presented in Figure 13 indicating the complex nature of the potential for ongoing elemental loading. In conjunction with Sebastiao et al's. 2017 study (Table S22 – SI), there is significant evidence of elemental loading in river waters where they meet ford crossings which are likely derived from vehicle wash-off and road surface runoff. What is not clear however, is the amount of airborne particulate matter that falls out into the freshwater system. Sahu and Elumalai (2017) identified a threefold increase in concentrations of Zn and Cu particulates on roadsides adjacent to traffic calming devices. It would seem sensible to assume that ford crossings act similarly to traffic calming devices as vehicles slow down on

547 approach. This could serve to exacerbate the metal loading from air-deposited particulate
548 matter, a subject for further study. Eight no. dustscan discs were placed around the study
549 site at locations SP1, SP3, SP4, SP7, SP9 and SP14. However, the results from this aspect
550 were inconclusive due to coverage of leaf litter and the loss of three of the discs , therefore,
551 the air monitoring element was omitted from the study

552 The Government of Western Australian Department for Water and Environmental
553 Regulation – Water Quality Protection Note (WQPN) identifies the need for careful selection
554 of crossing points over freshwaters. Appendix D of WQPN indicates a 75 m vegetated buffer
555 zone for bridge crossings to minimise contamination. However, no such protection
556 measures have been identified for ford crossings. The results from the chemical and
557 ecological analysis at Birchgrove and Twyford Lanes should be helpful when selecting new
558 crossing points over sensitive waters, especially in the light of increased infrastructural
559 pressures. An estimated critical distance of 150 m from the crossing is suggested to allow for
560 attenuation of metal concentrations to background levels. However, Figure 11 indicates that
561 this is dependent on the element concerned, for example the critical distance (the distance
562 for concentrations to attenuate toward backgrounds levels) for Zn was estimated to be 450
563 m as opposed to ca. 115 m for Cr. Pb fell between the two, with an estimated critical
564 distance of 180 m. Further study is also required to elucidate seasonal changes, i.e. changes
565 in winter include loss of vegetation cover reducing trapping of airborne particulates,
566 temperature changes and discharge in water during winter months may increase EC, alter
567 pH and decrease deposition, and finally, the addition of NaCl onto road surfaces in winter
568 which enters the water system will affect sorption and the reactional kinetics. Consideration
569 of sediment composition is also important. As indicated in Figures 10 and 12, bedload
570 sedimentology can, at first inspection, suggest misleading attenuation distances. Figure 12

571 shows lower concentrations of Cr and Pb in SP11, the sample point immediately
572 downstream of Ford 2, than those of SP2 – 10m downstream of Ford 2. An explanation of
573 the peak at SP12 is that the fine materials or the clays in SP12 adsorb the metals in question
574 more effectively than the hardstanding bed at SP11. Therefore, in future designs, although
575 hardstanding concrete may promote faster flow and lack of absorption, such designs may
576 skew attenuation data as contamination is mobilised only to be stored further downstream
577 in more favourable conditions.

578 Since vehicle manufacturing processes respond to the inevitable developments of new
579 technologies, so might the elemental contaminants introduced into the environment. As
580 seen with Pb, some contaminants have a long residence time and may persist long into the
581 future.

582 Although the study site is along relatively quiet stretches of rural roadways, it would be
583 expected that for more heavily trafficked areas the results would be magnified such as
584 identified in Sansalone et al. (1996), Davis et al. (2001), Loganathan et al. (2013) and
585 Sebbastiao et al. (2017). Therefore, the impact to ecology would be more significant. The
586 results highlight the importance of placing mitigation measures in areas where road-derived
587 sediments migrate into waterways. With reference to Wang et al. (2019), such strategies
588 should focus on very fine particles within road sediments and suspended in storm water, as
589 these are associated with higher accumulations of heavy metals, as indicated in discussed
590 earlier. Additionally, changes in Eh and pH associated with agricultural activities, sewage
591 discharges and rain, could release heavy metals, making them more bioavailable. Further
592 investigation into the speciation of identified elements would be required to elucidate this.
593 With the expected increase in vehicular usage likely to lead to an increasing demand for

more and better-quality roads, thought should be given to such strategies prior to any road construction.

The importance of understanding habitat fragmentation will increase, since mounting pressure on land use and the expected rise in vehicular traffic will continue to put strain on local environments. Although contamination into a river system via a road crossing (be it a ford or bridge) may only be localised, the impact is potentially catastrophic as “dead zones” may be created impeding important migratory routes. Additionally, habitat fragmentation will inevitably cause increased intra- and inter-species competition and alterations in trophic interactions (Wilson et al, 2015). As such, understanding the chemical and biological dynamics of river crossings will help planners give further thoughts to placement and mitigation of such crossings, particularly in sensitive areas.

6.0 CONCLUSIONS

- Heavy metals Pb, Cr and Zn in sediments, and only Zn in water, show a marked increase in concentration at the closest downstream side of ford crossings. The concentrations of metals subsequently decrease with distance downstream, only to spike again at the next ford.
- Attenuation distances vary. For instance, in this study, Zn returns to background range by ca. 450 m, but the critical distances for Cr and Pb are significantly shorter at ca. 106 and ca. 180 m respectively.
- As a result of the increased heavy metal loading, biodiversity and the presence of sensitive taxa was significantly reduced at positions located near ford crossings.
- No metal concentration in this study exceeded appropriate guidance values for human health.

- The results of chemical and ecological analysis should be considered during selection and design of new crossings over sensitive waters, especially in the light of increased infrastructural pressures.

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Determinand	Result Range (mg/kg)	Results Average (mg/kg)	Average concentration of control Samples (mg/kg)	Est. Background Geochemistry Value (mg/kg)	SGV (mg/kg)	C4SL/SSAC	Human Health Exceedances
Arsenic	3.0 – 13.8	6.28	2.45	<15		32	N
Cadmium	<0.5 – 0.6	-	Not tested [‡]	1.8		5	N
Chromium	4.0 – 41.2	18.78	7.47	60 – 90		1950	N
Copper	<5.0 – 11.7	7.34	5.2	-		5203	N
Lead	7.2– 27.3	16	6.2	<100	200		N
Magnesium	192 – 6200	1186	297				-
Manganese	137 - 2370	773	57.1				-
Mercury	<0.5	BLOD*	Not Tested [‡]			168	N
Molybdenum	<0.5	BLOD*	Not Tested [‡]				
Nickel	5.9 – 40.1	17.34	BLOD*	15 - 30		533	N
Zinc	31.6 – 155.7	95.64	13.37			19400	N
Hexavalent Chromium	<0.8	BLOD*	Not Tested [‡]		21	21	N

*BLOD = Below Limit of Detection

‡ Background levels not tested in control sample because sample point results were below detection limit

Table 1: Average Results of Concentrations within Bed Sediments compared to Human Health Criteria

Determinand	Phytotoxicity GAC (mg/kg)			GAC Exceedances
	pH <6.0	pH 6.0-7.0	pH >7.0	
Zinc	200	200	300	No
Copper	100	135	200	No
Nickel	60	75	110	No

Table 2: Summary of Plant Phytotoxicity Assessment No exceedances of Generic Assessment Criteria for Zn, Cu and Ni were identified within the stream sediments.

Figure 1: The study area along with photo-view of fords and sampling sites across the fords. The area occupies a section of a tributary of the River Ouse, located in a shallow valley of forest and agrarian land in Sussex. The tributary is joined (immediately upstream of the fords) by two minor channels (Minor Stream 1 & 2), one at each ford. Another minor stream (Minor Stream 3), cutting into the roadside, runs alongside Birchgrove Lane joining the tributary immediately downstream of Ford 2. Both, Minor Streams 1 & 3, are ephemeral and were dry on several visits. Ref: Edina Digimap (2018). *Colour Raster: Sussex 1:50,000 [PDF] OS Roam Series*. Retrieved from <http://digimap.edina.ac.uk/roam/os>

Figure 2: A conceptual site model of the study area illustrating the risk rendering from potential source-pathways-receptor interaction. Topography and land use make the tributary and stream network vulnerable to contamination from various sources.

Figure 3: Metal Concentrations in Water: Zn, Mg and Mn are the elements that were found to occur at high concentrations. The arrows at the bottom of each graph represent where the ford locations are, samples 4 and 11 are taken immediately downstream of Ford 1 and Ford 2 respectively

	Control Samples		Twyford Lane (Ford 1)		Birch Grove Lane (Ford 2)	
	Range (mg/kg)	Average (mg/kg)	Range (mg/kg)	Average (mg/kg)	Range (mg/kg)	Average (mg/kg)
Arsenic	1.8 – 3.8	2.45	3.7 – 11.2	6.7	3.0 – 8.0	5.16
Chromium	<5 – 11.1	7.47	23.4 – 34.5	28.5	33.2 – 41.2	37.0
Copper	<5 – 6.2	5.2	<5 – 8.1	6.43	6.3 – 11.7	9.4
Lead	5.2 – 7.2	6.2	14.2 – 18.2	16.48	19.8 – 27.3	23.8
Magnesium	295 - 356	297	192 - 1010	496	1630–6200	3218
Manganese	40.5 – 153	57.1	138 - 567	354	561 – 1790	1003
Nickel	<5	-	19.6 – 35.0	24.97	8.2 – 12.7	11.65
Zinc	11.5 – 19.3	13.37	98.6 – 155.7	134.28	98.8 - 144.4	121.13

Figure 4: Comparison of Background and Ford Contaminant Concentrations (Base 10 Log Scale). The control sample results were compared to the sediment samples taken from within the stream. The concentrations of elements shown above highlight that elemental concentrations are consistently elevated in the ford crossings compared to concentrations within the control samples.

Figure 5: Contaminants Vs Road Distance and Ford Distance. An inverse association between the concentration of elements and distances from the road and fords was identified.

Figure 6: Spatial Distributions of Chromium, Lead and Zinc: Note: The Scale is Comparable to background Readings Identified in Fig. 4. The spatial diagram shows highest concentrations for Cr, Pb and Zn immediately downstream of the ford crossings.

Figure 7: BMWP Score Vs Road Distance (a) and BMWP Score Vs Ford Distance (b) Spatial Distribution of BMWP Scores (c): 7a): A strong negative correlation between BMWP score and distance from the road is identified by an R^2 value of 0.56. 7b): A significant correlation between increased distance from the ford crossing and increased BMWP score with an R^2 value of 0.69, further backed up by a Pearson's coefficient of 0.72. 7c): In general, BMWP scores are lower ("moderate" to "poor") in, and in proximity to, the ford crossings. "Very Good" scores are recorded with increased distances from the ford.

Family

Ephemeroptera (Mayfly)

Plecoptera (Stonefly)

Trichoptera (Caddisfly)

Figure 8 Invertebrate Abundance. Ephemeroptera and plecopteran are noted as being rare in Ford 1 and absent in Ford 2. Conversely, Ephemeroptera are noted as being frequent or abundant in sampling positions away from the ford crossings and road, whilst Plecoptera are recorded as being occasional further from the fords and roads. A slightly different pattern was observed for the less sensitive Trichoptera, which was found to be “occasional at Ford 1 and “frequent” at Ford 2.

Figure 9: Metals Vs BMWP Scores: Cr, Pb and Zn and to a lesser extent Cu, Ni, Mn and Mg, have a negative influence on BMWP scores.

Figure 10: Relationship Between Particle Size, SOM and Elemental Concentration

Figure 11: Estimated downstream attenuation distances (i.e. distance to return to background range) for Zn, Cr and Pb. Zn is estimated to not return within the normal background range until ca.450m downstream of a ford. The critical distances for Cr and Pb are ca.106m & ca.180m respectively.

Figure 12: Concentrations of Pb and Cr at sample points near to Ford 2: Higher concentrations of some elements were recorded at SP12 (10m downstream from Ford 2) than they were at SP11 (located immediately downstream of Ford 2). For example, Pb and Cr concentrations at SP11 were 19.8mg/kg and 36.7mg/kg respectively. Whereas at SP12, where slightly lower concentrations would be expected, concentrations were 24.3mg/kg and 41.2mg/kg respectively. The concentrations then regress further downstream at SP13 and SP14.

Figure 13: Overall Conceptual Contamination Model: Illustration the mechanism of contamination by vehicular wash off and influence of topography and road wash off, immediately after rainfall/precipitation event.