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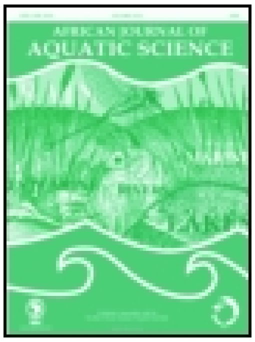

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Effect of drift sampler exposure time and net mesh size on invertebrate drift density in the Njoro River, Kenya

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Although invertebrate drift is an important ecological process in lotic ecosystems, very little is known about it in Kenyan rivers. The primary aim of this study was to investigate the effect of driftnet mesh size and exposure duration on drift density in 2017. Drift samples were dominated by Chironomidae, Baetidae, Simuliidae, Caenidae and Culicidae. The 100 µm mesh driftnet had the highest mean invertebrate density, followed by the 250 µm and 500 µm nets. Invertebrate drift densities decreased with increased exposure time. This study demonstrates that sampler mesh size and exposure time should be taken into account when characterising invertebrate drift in streams. Future studies should consider sampling different biotopes and during different seasons.

Keywords: lotic ecosystems, macroinvertebrates, Nakuru County, riffle habitats

Online Supplementary Material: Effect of drift sampler exposure time and net mesh size on invertebrate drift density in the Njoro River, Kenya is available at doi.org/10.2989/16085914.2018.1465394

Introduction

Intensive investigations of invertebrate drift (Müller 1954; Waters 1965; Brittain and Eikeland 1988; Naman et al. 2016) have resulted in different hypotheses regarding invertebrate drift. According to Müller (1954) and others (Williams and Williams 1993; Pachepsky et al. 2005), invertebrates drift in reaction to living in lotic ecosystems by developing upstream flight, while others (Waters 1966; Anholt 1995; Turner and Williams 2000) hypothesised that invertebrate drift was dependent on the extent to which the carrying capacity of a lotic system is overloaded, thus providing a way of removing surplus production. It is generally recognised that invertebrate drift is a fundamental ecological process that plays a crucial role in recolonization of perturbed areas, as well as being a food source for drift feeding fish, and helps stream invertebrates to avoid predators (Waters 1964; Brittain and Eikeland 1988; Naman et al. 2016). Invertebrate drift in lotic systems typically mirrors the benthic community, providing a valuable means of assessing benthic invertebrates (Koetsier et al. 1996; Pringle and Ramírez 1998; Shearer et al. 2003).

Invertebrates may drift, because of accidental dislodgement from the substratum, the presence of toxicants and sediment input, as well as changes in food resources, physical habitat structure, temperature and photoperiod (Brittain and Eikeland 1988; Naman et al. 2016; Katano et al. 2017). Drift can also be influenced by the presence of predators, high benthic densities, riparian land use activities, seasonality, or physical disturbances within the stream-bed (Ormerod et al. 2004; Naman et al. 2016; Weber et al. 2017). In a study evaluating invertebrate drift in adjacent sand-bed and riffle habitats, Gibbins et al. (2010) demonstrated that although water velocity did not differ between the habitats, deposited and suspended sediment concentrations were higher in the sand-bed habitat. Drift densities were considerably higher in the sandbed compared with the riffle suggesting that, based on the composition of invertebrates in benthic and drift samples, the riffle contributed to the higher drift densities in the downstream sand-bed habitat. Leung et al. (2009) in another study on the effect of habitat on invertebrate drift in Canada, found no relationship between invertebrate drift density and water velocity within individual pools or riffles, suggesting that turbulence and short distances between high-and low velocity habitats minimise changes in drift densities through settlement of invertebrates in areas with low velocity. Structurally complex riffles (Brown and Brussock 1991; Degani et al. 1993; Capderrey et al. 2013) provide invertebrates with refuge areas against predators and offer great food supply to benthic communities (Brown and Brussock 1991; Mathooko 1995; Wallace and Webster 1996). Hansen and Closs (2007) found that invertebrate drift densities increased with riffle area and length, while Grzybkowska et al. (2004) found that invertebrate drift densities were highest in riffles than in pools.

Research on invertebrate drift to date has largely focused on temporal patterns, effect of physico-chemical variables, such as light intensity, water temperature, sediment, habitat type, predator presence, life stage, disturbance and competition for resources (Brittain and Eikeland 1988; Naman et al. 2016; Béjar et al. 2017; Weber et al. 2017).

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Measurement of invertebrate drift composition and density is, however, also influenced by the duration of sampling, and the mesh size of nets (Slack et al. 1991; Culp et al. 1994; Leung et al. 2009). Slack et al. (1991) evaluating the effect of net mesh-size on observed invertebrate drift composition in a USA mountain stream observed a general trend of increasing abundance and number of invertebrate taxa with decreasing mesh size. Comparatively higher numbers of invertebrates occurred in the nets with a larger mesh size at night than during the day (see also diel periodicity in Flecker 1992; Ramírez and Pringle 1998; Huhta et al. 2000). Generally, driftnet mesh size and exposure time depend on the size and density of the organisms studied and the amount of suspended coarse sediment and organic matter (Muehlbauer et al. 2017). However, drift sampler exposure times vary greatly between studies, ranging from a few minutes to hours, using varied net mesh sizes (Flecker 1992; Kerby et al. 1995; Kennedy et al. 2014).

Despite the ecological importance of invertebrate drift and international interest in its dynamics, comparatively few studies have assessed this phenomenon in Kenyan rivers (e.g. Mathooko and Mavuti 1992), where most studies have primarily focused on benthic communities (e.g. Mathooko 2000; M'Erimba et al. 2014; Mbaka et al. 2016). The main aim of this study is to evaluate the effect of driftnet mesh size and exposure time on invertebrate drift densities in the Njoro River, Kenya. It is hypothesised that drift sampler exposure time and net mesh size have an effect on invertebrate drift densities and composition.

Materials and methods

Study area and site

The study site was located on a reach of the Njoro River in Nakuru County, Kenya (Figure 1). This 55 km long second order stream rises in the eastern Mau hills at an elevation of 2 880 m asl, draining a 250 km² catchment (Osano 2015). The dry season typically lasts from December to March and the wet season from April to November (Mbaka et al. 2014). The soils are predominantly clay loam, though silt clay is common near Lake Nakuru (Karanja et al. 1986). The two main vegetation types include the montane *Juniperus procera-Olea europaea* subsp. *africana* and submontane *Acacia abyssinica* forests (Mathooko and Kariuki 2000).

The study was conducted in a 20 m long, 5.5 m wide riffle (Figure S1) in the middle reaches of the Njoro River at 2 263 m asl. The right and left banks of this river section border Njokerio trading centre and Egerton University, respectively. Human-related physical perturbations at the site were low, except for small-scale farming on the right bank. There was a 70% canopy cover of *Syzygium cordatum, Euclea* sp.*, Juniperus* sp. and *Maytenus senegalensis,* the most common riparian plants. The riffle habitat was composed of 90% bedrock, 5% cobbles, and 5% silt and sand. Additional details on physical and chemical characteristics of the study site are presented in Table 1.

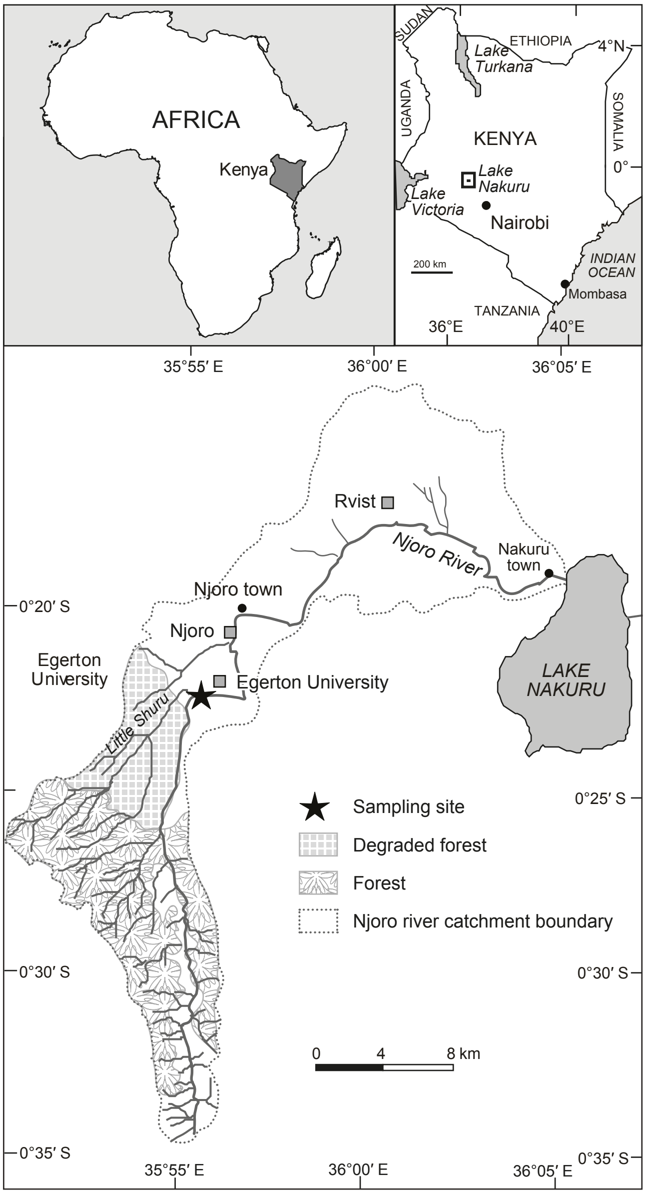

Figure 1: Location of sampling site in the Njoro River

Table 1: Mean ± SE and range of chemical and physical variables in the Njoro River at the study site in 2017

Variable	Mean (±SE)	Range
Temperature (°C)	15.7 (0.3)	13.8–18.7
Conductivity (μS cm⁻¹)	202.4 (17.4)	146.4–295.0
Dissolved oxygen (mg l⁻¹)	7.8 (0.1)	7.0–8.9
Oxygen saturation (%)	92.2 (1.6)	80.0–103.8
Turbidity (NTU)	20.7 (0.6)	17.6–25.9
Velocity (m s⁻¹)	0.8 (0.1)	0.3–1.3
Discharge (m³ s⁻¹)	0.4 (0.1)	0.1–1.0

Water chemical and physical characteristics

Sampling was conducted seven times between 23 January and 8 March, 2017 (Table S1). Water temperature, pH, dissolved oxygen, oxygen saturation, turbidity and electrical conductivity were measured *in-situ* using portable sensors. Electrical conductivity, pH and temperature were measured using a HACH HQ 40d meter. Turbidity was measured using a HACH HQ 11d meter and oxygen concentration using a HACH HQ 30d meter. Water velocity was measured at 60% of water depth using a portable automatic flow meter (Flo-Mate, model 2000, Marsh McBirney). The composition of river substrates and canopy cover were assessed visually.

Drift sampling and invertebrates processing

Drift sampling was conducted between 10:00 and 16:00 h during the day, using three drift samplers fitted with 100 μm, 250 μm and 500 μm mesh nets, respectively. The samplers were placed side by side (right bank, midstream, left bank) in the riffle facing the upstream (Figure S1). The samplers were exposed for 5, 10, 15, 20, 25 and 120 minutes (Table S2). The procedure was repeated for three consecutive days, with the position of the three drift samplers being interchanged, ensuring that each drift sampler was accorded an equal chance of being in each of the three riffle habitat positions. The mean current velocity of water passing through the mouth of each sampler was measured at 60% of the water depth. Invertebrates collected in the cup at the rear end of the driftnet were placed in labelled polythene bags, preserved with 4% formalin. In the laboratory the samples were sieved to remove coarse organic matter and sediment before the invertebrates were sorted under a dissecting microscope, enumerated and identified to the lowest possible taxonomic level following Gerber and Gabriel (2002), de Moor et al. (2003) and Day et al. (2003). All invertebrates present in the samples were considered, and drift density was expressed as ind. m^{-3}.

Data analysis

The effect of driftnet mesh and exposure on invertebrate drift density was tested using Linear Mixed-Effect Models (LMM), with mesh size and exposure time as fixed factors, mesh size as an interaction term with exposure time and sampler position as a random factor. The *p*-values were corrected in multiple tests following Holm (1979) and data distribution was evaluated following Zuur et al. (2009) and Ghasemi and Zahediasl (2012). Post-hoc evaluations were made using Tukey contrasts (Hothorn et al. 2008). Data analysis was performed using the R statistical Package (R Development Core Team 2015).

Results

Chironomidae, Baetidae, Simuliidae, Caenidae and Culicidae were the most abundant taxa in the Njoro River drift samples (Table S3). The Chironomidae had the highest densities in all the three net sizes, followed by Baetidae, Simuliidae, Caenidae and Culicidae. In total, 23 invertebrate taxa were recorded in the drift samples (Table S4). The highest mean invertebrate drift density (1.8 ± 0.3 ind. m^{-3}) was recorded for the drift sampler with the 100 μm driftnet, followed by the 250 μm driftnet (1.7 ± 0.3 ind. m^{-3}) and 500 μm driftnet (0.7 ± 0.1 ind. m^{-3}) (Figure 2). Mesh size had a statistically significant effect ($F_{2,53}$ = 4.9, p = 0.01) on invertebrate drift densities, whereas Tukey contrasts indicated a statistically significant difference ($p < 0.05$) between the densities of invertebrates collected in 100 μm and 500 μm nets.

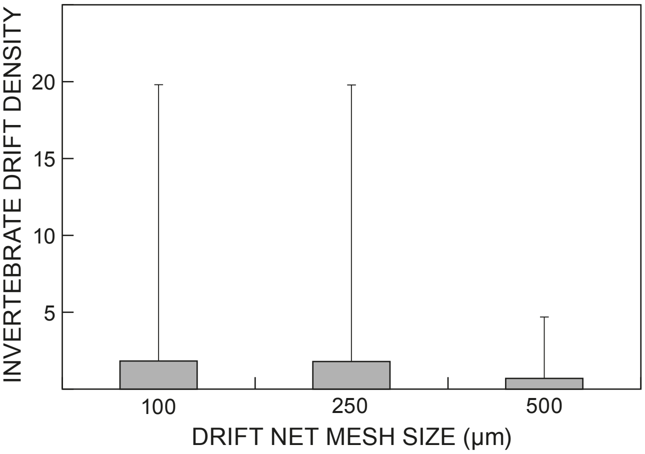

Figure 2: Mean and SD (vertical bars) densities (ind. m^{-3}) of invertebrates collected in 100 μm, 250 μm and 500 μm driftnets in the Njoro River in 2017

In general, invertebrate drift densities decreased with increased exposure time (Figure 3). The highest mean invertebrate drift density (3.9 ± 1.6 ind. m^{-3}) was recorded for the 5-minute exposure using the 100 μm driftnet, whereas the lowest mean invertebrate drift density (0.2 ± 0.05 ind. m^{-3}) was recorded for the 120-minute exposure 500 μm mesh driftnet (Figure 3). Drift sampler exposure time had a statistically significant effect ($F_{5,53}$ = 3.4, p = 0.01) on invertebrate drift densities and Tukey contrasts indicated that there was a significant difference (p = 0.007) between the 5-minute and 120-minute exposure times. There was an insignificant interaction term between mesh size and exposure time ($F_{10,53}$ = 0.7, p = 0.73). The highest mean invertebrate drift density (3.8 ± 0.9 ind. m^{-3}) occurred for the 100 μm mesh net at the left bank position, whereas the lowest mean invertebrate drift density (0.5 ± 0.06 ind. m^{-3}) occurred for the 500 μm mesh net at the midstream sampling position (Figure 4).

Discussion

Mean invertebrate drift density decreased with increasing net mesh size, demonstrating that fine-meshed nets have the ability to collect more invertebrates than coarse-meshed nets. This difference can be attributed to the loss of small invertebrates passing through coarse-meshed nets. Mbaka et al. (2016) demonstrated that coarse-meshed sieves (500 μm) resulted in the exclusion of meiofauna from samples and had a significant effect on mean invertebrate density (See also Hwang et al. 2007; Pinna et al. 2014; Hartwell and Fukuyuma 2015). In a study assessing the contribution of meiofauna to invertebrate drift, Perić et al. (2014) found that meiofauna constituted 35% of total invertebrate drift density. Given that small-sized fauna are likely to be lost from drift samples when using coarse-meshed nets, it is important to use fine-meshed nets when characterising invertebrates in stream ecosystems where the existing taxa are unknown.

The general tendency to use fine-meshed driftnets (Slack et al. 1991) is evident from previous studies on the effect

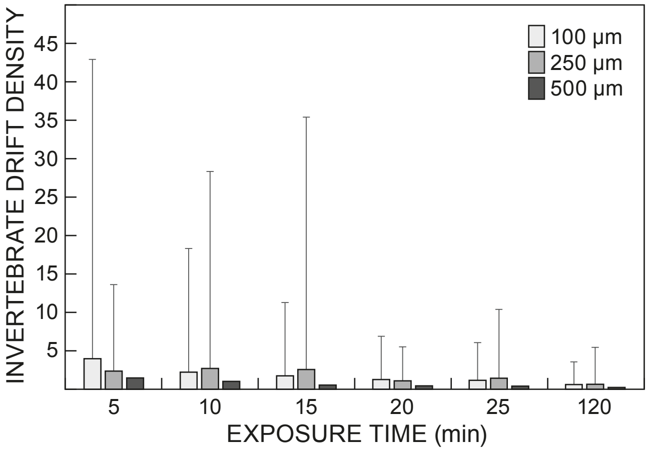

Figure 3: Mean and SD (vertical bars) drift densities (ind. m^{-3}) of invertebrates sampled during different exposures times (5–120 min) in the Njoro River in 2017

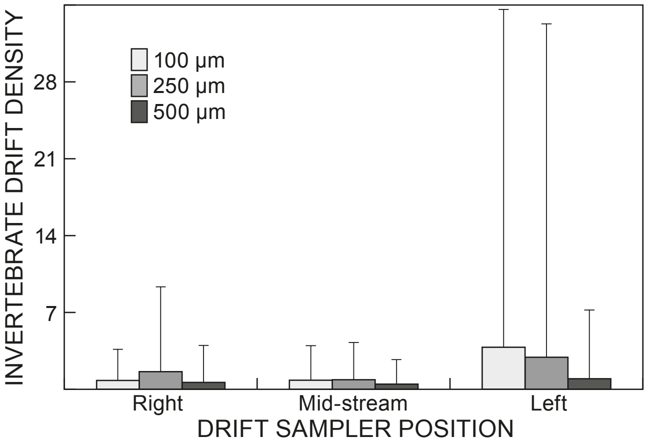

Figure 4: Mean and SD (vertical bars) densities (ind. m^{-3}) of invertebrates that drifted at the right bank, midstream and left bank positions of the sampled riffle biotope in the Njoro River in 2017

of mesh size on invertebrate drift density (Clifford 1972; Ferrington 1984; Slack et al. 1991). Although fine-meshed nets retain the small invertebrates, sample sorting time and rate of net clogging may possibly lead to underestimation of invertebrate drift density and composition. Coarse-meshed driftnets may, however, be more appropriate if the objective is to analyse macroinvertebrates. It is fundamental to determine the optimum exposure time in order to obtain representative samples and simultaneously avoid clogging when using fine-meshed nets.

Mean invertebrate drift densities decreased with increased exposure time. The 500 μm driftnet had the lowest mean density during the 120-minute exposure time. The optimum exposure time in this study was 5 minutes using the 100 μm driftnet. The invertebrate drift density consistently decreased with increasing time using the 100 μm driftnet, possibly because of clogging (e.g. Slack et al. 1991; Muehlbauer et al. 2017), whereas drift densities did not vary much from 15- to 120-minute exposure time for the 500 μm driftnet (Figure 3). This implies that the fine-meshed driftnet was more appropriate for sampling invertebrate drift in the Njoro River within the shorter exposure time frame.

Modification of driftnet filtering efficiency because of clogging results in reduced net entrance velocities, resulting in incorrect calculation of sampled water volume, and consequently the invertebrate drift density (Faulkner and Copp 2001). In a meta-analysis of 77 studies on the effect of driftnet clogging on drift concentrations, Muehlbauer et al. (2017) found that driftnets clog in a non-linear fashion over time, and that coarse suspended solids and net mesh size have a strong impact on clogging rates and the resultant drift data.

Given that linear models are typically used to derive the total volume of water filtered over a given exposure time, the most appropriate model (e.g. inverse exponential, logistic) should be considered where clogging occurs (Muehlbauer et al. 2017). The non-linear fashion in which driftnets clog also suggests that the typically used method of calculating average water velocity from measurements taken only at the start and end of sampling could result in considerable errors in density values. Despite this, the optimum sampling time for a driftnet of a given mesh size and the suspended material in a given lotic ecosystem are rarely taken into consideration. Studies can overcome the problem of fine-meshed nets clogging by reducing the exposure time or by modifying the sampling methods to avoid clogging if the size range of organisms does not lead to gross underestimation of drift densities and composition. Some smaller invertebrates, such as water mites, ostracods, etc., may be lost in large numbers if the mesh nets used have larger (>500 μm) pore openings.

Although exposure time and mesh size had a significant effect on drift density, there was no significant interaction between the two factors, suggesting that the effect of mesh size on drift density is not significantly modified (Baron and Kenny 1986; Aguinis and Gottfredson 2010) by exposure time, and vice versa. These two factors can thus be regarded as independent factors influencing drift.

The mean invertebrate drift densities (0.7–1.8 ind. m^{-3}) recorded in this study are within the range (0.5–3 ind. m^{-3}) of those measured in other previous drift studies (Boyero and Bosch 2002; Bruno et al. 2010; Astakhov and Bogatov 2014). The high mean densities of individual taxa, such as Chironomidae, Baetidae, Simuliidae, Caenidae and Culicidae in this study have also been reported in other invertebrate studies (Pringle and Ramírez 1998; Bruno et al. 2010; Astakhov and Bogatov 2014). Although this study did not explicitly link drift densities to level of human disturbance, invertebrate drift has crucial ecological roles and is important in river biomonitoring (Pringle and Ramírez 1998; Gimenez et al. 2015; Naman et al. 2016). Pringle and Ramírez (1998) found that both benthic and drift sampling techniques indicated Chironomidae and Ephemeroptera as the dominant insect groups in a tropical stream. Chironomidae dominated the disturbed stream draining agricultural areas. Whereas insects were dominant (>90%) in benthic samples, larval Atyidae were dominant (>50%) in drift samples, demonstrating the importance of routinely measuring invertebrate drift. Gimenez et al. (2015) found a higher taxon diversity in a rural stream where sensitive taxa, such as

Ephemeroptera, Plecoptera, Trichoptera and Coleoptera were dominant, while high densities of Chironomidae drifted in an urban stream. The composition of invertebrate drift therefore primarily reflects the benthic fauna and conditions. The invertebrate taxa recorded in the current study have also been found to be a major component in benthos (e.g. Mathooko 2001; M'Erimba et al. 2014; Mbaka et al. 2014).

The highest mean invertebrate drift density was recorded for the 100 µm drift sampler placed at the left side of the riffle. This can be attributed to the large hydraulic disturbance (Figure S2) experienced by invertebrates, making them more susceptible to drift. Large hydraulic disturbance in the benthic zone may also increase the drift of benthic organic matter greatly, and consequently the drift rate of invertebrates, because of reduction of refugia, food and attachment surfaces.

Conclusions

Driftnet mesh size and exposure time had significant independent effects on invertebrate drift density. Accordingly, mesh size and exposure duration are important factors to consider when sampling drift in streams. Fine-meshed (100 µm) driftnets are the most appropriate for invertebrate drift density sampling, albeit with modified exposure time. Future studies should consider sampling different seasons, habitats and potential for drift to identify disturbance in streams.

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References

Aguinis H, Gottfredson RK. 2010. Best-practice recommendation for estimating interaction effects using moderated multiple regression. *Journal of Organizational Behavior* 31: 776–786.

Anholt BR. 1995. Density dependence resolves the stream drift paradox. *Ecology* 76: 2235–2239.

Astakhov MV, Bogatov VV. 2014. Vertical redistribution of drifting benthic invertebrates in the Kedrovaya River, Primorsky Region of Russia. *Open Journal of Ecology* 4: 53–59.

Baron RM, Kenny DA. 1986. The moderator-mediator variable distinction in social psychological research: conceptual, strategic, and statistical considerations. *Journal of Personality and Social Psychology* 51: 1173–1182.

Béjar M, Gibbins CN, Vericat D, Battalla RJ. 2017. Effects of suspended sediment transport on invertebrate drift. *River Research and Applications* 33: 1655–1666.

Boyero L, Bosch J. 2002. Spatial and temporal variation of macroinvertebrate drift in two Neotropical streams. *Biotropica* 34: 567–574.

Brittain JE, Eikeland TJ. 1988. Invertebrate drift-a review. *Hydrobiologia* 166: 77–93.

Brown AV, Brussock PP. 1991. Comparisons of benthic invertebrates between riffles and pools. *Hydrobiologia* 220: 99–108.

Bruno MC, Maiolini B, Carolli M, Silveri L. 2010. Short time-scale impacts of hydropeaking on benthic invertebrates in an Alpine stream (Trentino, Italy). *Limnologica* 40: 281–290.

Capderrey C, Datry T, Foulquier A, Claret C, Malard F. 2013. Invertebrate distribution across nested geomorphic features in braided-river landscapes. *Freshwater Science* 32: 1188–1204.

Clifford 1972. A year's study of the drifting organisms in a brown-water stream of Alberta, Canada. *Canadian Journal of Zoology* 50: 975–983.

Culp JM, Scrimgeour GJ, Beers CE. 1994. The effect of sample duration on the quantification of stream drift. *Freshwater Biology* 31: 165–173.

Day JA, Harrison AD, de Moor IJ. 2003. Guide to the freshwater invertebrates of southern Africa. Volume 9: Diptera.WRC Report No. TT 201/02. Pretoria: Water Research Commission.

Degani G, Herbst GN, Ortal R, Bromley HJ, Levanon D, Netzer Y, Harari N, Glazman H. 1993. Relationship between current velocity, depth and the invertebrate community in a stable river system. *Hydrobiologia* 263: 163–172.

de Moor IJ, Day JA, de Moor FC. 2003. Guide to the freshwater invertebrates of southern Africa. Volume 8: Insecta II. WRC Report No. TT 214/03, Pretoria: Water Research Commission.

Faulkner H, Copp GH. 2001. A model for accurate drift estimation in streams. *Freshwater Biology* 46: 723–733.

Ferrington LC. 1984. Drift dynamics of Chironomidae larvae: I. Preliminary results and discussion of importance of mesh size and level of taxonomic identification in resolving Chironomid diel drift patterns. *Hydrobiologia* 114: 215–227.

Flecker AS. 1992. Fish predation and the evolution of invertebrate drift periodicity: evidence from Neotropical streams. *Ecology* 73: 438–448.

Gerber A, Gabriel MJM. 2002. *Aquatic invertebrates of South African rivers: field guide.* Pretoria: Institute for Water Quality Studies, Department of Water Affairs and Forestry.

Ghasemi A, Zahediasl S. 2012. Normality tests for statistical analysis: a guide for non-statisticians. *International Journal of Endocrinology and Metabolism* 10: 486–489.

Gibbins CN, Vericat D, Batalla RJ. 2010. Relations between invertebrate drift and flow velocity in sand-bed and riffle habitats and the limits imposed by substrate stability and benthic density. *Journal of the North American Benthological Society* 29: 945–958.

Gimenez BCG, Lansac-Toh FA, Higuti J. 2015. Effect of land use on the composition, diversity and abundance of insects drifting in neotropical streams. *Brazilian Journal of Biology* 75: S52–S59.

Grzybkowska M, Dukowska M, Figiel K, Szczerkowska E, Tszydel M. 2004. Dynamics of macroinvertebrate drift in a lowland river. *Zoologica Poloniae* 49: 111–127.

Hansen EA, Closs GP. 2007. Temporal consistency in the long-term spatial distribution of macroinvertebrate drift along a stream reach. *Hydrobiologia* 575: 361–371.

Hartwell SI, Fukuyuma AK. 2015. The effects of sieve size on benthic community composition analysis. *Journal of Coastal Research* 31: 1531–1536.

Holm S. 1979. A simple sequentially rejective multiple test procedure. *Scandinavian Journal of Statistics* 6: 65–70.

Hothorn T, Bretz F, Westfall P. 2008. Simultaneous inference in general parametric models. *Biometric Journal* 50: 346–363.

Huhta A, Muotka T, Tikkanen P. 2000. Nocturnal drift of mayfly nymphs as a post-contact anti-predator mechanism. *Freshwater Biology* 45: 33–42.

Hwang J, Kumar R, Dahms H, Tseng L, Chen Q. 2007. Mesh size affects abundance estimates of *Oithona* spp. (Copepoda, Cyclopoida). *Crustaceana* 80: 827–837.

Karanja AK, China SS, Kundu P. 1986. The influence of land use on Njoro River catchment between 1975 and 1985. In: Thomas DB, Biamah EK, Kilewe AM, Lundgren L and Mochoge BO (Eds), *Proceedings of the third national workshop on soil and water conservation, 16–19 September, 1986*. University of Nairobi Press, Nairobi. pp 40–49.

Katano I, Mitsuhashi H, Doi H, Isobe Y, Oishi T. 2017. Distribution and drift dispersal dynamics of a caddisfly grazer in response to resource abundance and its ontogeny. *Royal Society Open Science* 4: 160732.

Kerby BM, Bunn SE, Hughes JM. 1995. Factors influencing invertebrate drift in small forest streams, south-eastern Queensland. *Marine and Freshwater Research* 46: 1101–1108.

Kennedy TA, Yackulic CB, Cross WF, Grams PE, Yard MD, Copp AJ. 2014. The relation between invertebrate drift and two primary controls, discharge and benthic densities, in a large regulated river. *Freshwater Biology* 59: 557–572.

Koetsier P, Minshall WG, Robinson CT. 1996. Benthos and macroinvertebrate drift in six streams differing in alkalinity. *Hydrobiologia* 317: 41–49.

Leung ES, Rosenfeld JS, Bernhardt JR. 2009. Habitat effects on invertebrate drift in a small trout stream: implications for prey availability to drift-feeding fish. *Hydrobiologia* 623: 113–125.

Mathooko JM. 1995. The retention of plant coarse particulate organic matter (CPOM) at the surface of the wet-store and dry-store zones of the Njoro River, Kenya. *African Journal of Ecology* 33: 151–159.

Mathooko JM. 2000. Artificial physical disturbances on stream-bed sites of a Kenyan mountain stream: effects of inter-disturbance intervals on mayfly densities. *Limnologica* 30: 331–339.

Mathooko JM. 2001. Disturbance of a Kenyan Rift Valley stream by the daily activities of local people and their livestock. *Hydrobiologia* 458: 131–139.

Mathooko JM, Kariuki ST. 2000. Disturbances and species distribution of the riparian vegetation of a Rift Valley stream. *African Journal of Ecology* 38: 123–129.

Mathooko JM, Mavuti KM. 1992. Composition and seasonality of benthic invertebrates, and drift in the Naro Moru River, Kenya. *Hydrobiologia* 232: 47–56.

Mbaka JG, M'Erimba CM, Thiongo HK, Mathooko JM. 2014. Water and habitat quality assessment in the Honi and Naro Moru Rivers, Kenya, using benthic macroinvertebrate assemblages and qualitative habitat scores. *African Journal of Aquatic Science* 39: 361–368.

Mbaka JG, M'Erimba CM, Karanja HT, Mathooko JM, Mwaniki MW. 2016. Does the exclusion of meiofauna affect the estimation of biotic indices using stream invertebrates? *African Zoology* 51: 91–97.

M'Erimba CM, Mathooko JM, Ouma KO. 2014. Macroinvertebrate distributions in relation to human and animal-induced physical disturbance on the sediment surface in two Kenya tropical Rift Valley streams. *African Journal of Aquatic Science* 39: 337–346.

Muehlbauer JD, Kennedy TA, Copp AJ, Sabol T. 2017. Deleterious effects of net clogging on the quantification of stream drift. *Canadian Journal of Fisheries and Aquatic Sciences* 74: 1041–1048.

Müller K. 1954. Investigations on the organic drift in North Swedish streams. Unpublished report, Institute of Freshwater Research, Drottningholm 34: 133–148.

Naman SM, Rosenfeld JS, Richardson JS. 2016. Causes and consequences of invertebrate drift in running waters: from individuals to populations and trophic fluxes. *Canadian Journal of Fisheries and Aquatic Sciences* 73: 1292–1305.

Ormerod SJ, Jones ME, Jones MC, Phillips DR. 2004. The effects of riparian forestry on invertebrate drift and brown trout in upland streams of contrasting acidity. *Hydrology and Earth System Sciences* 8: 578–588.

Osano PO. 2015. Morphometric characterisation and hydrological assessments of River Njoro watershed using system for Automated Geoscientific Analysis (SAGA) and Shuttle Radar Topographic Mission (SRTM) digital elevation model. *International Journal of Advances in Remote Sensing and GIS* 4: 37–44.

Pachepsky E, Lutscher F, Nisbet RM, Lewis MA. 2005. Persistence, spread and the drift paradox. *Theoretical Population Biology* 67: 61–73.

Perić MS, Dražina T, Špoljar M, Radanović I, Primic B, Habdija I. 2014. Meiofauna constitute a considerable portion of invertebrate drift among moss-rich patches within a karst hydrosystem. *Biologia* 69: 363–380.

Perić MS, Robinson CT. 2015. Spatio-temporal shifts of macroinvertebrate drift and benthos in headwaters of a retreating glacier. *Hydrobiologia* 751: 25–41.

Pinna M, Marini G, Mancinelli G, Basset A. 2014. Influence of sampling effort on ecological descriptors and indicators in perturbed and unperturbed conditions: A study case using benthic macroinvertebrates in Mediterranean transitional waters. *Ecological Indicators* 37: 27–39.

Pringle CM, Ramírez A. 1998. Use of both benthic and drift sampling techniques to assess tropical stream invertebrate communities along an altitudinal gradient, Costa Rica. *Freshwater Biology* 39: 359–373.

R Development Core Team (2015). *A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. http://www. R-project.org/. [Accessed 17 May, 2017].

Ramírez A, Pringle CM. 1998. Invertebrate drift and benthic community dynamics in a lowland neotropical stream, Costa Rica. *Hydrobiologia* 386: 19–26.

Shearer KA, Stark JD, Hayes JW, Young RG. 2003. Relationships between drifting and benthic invertebrates in three New Zealand rivers: implications for drift-feeding fish. *New Zealand Journal of Marine and Freshwater Research* 37: 809–820.

Siler ER, Wallace JB, Eggert SL. 2001. Long-term effects of resource limitation on stream invertebrate drift. *Canadian Journal of Fisheries and Aquatic Sciences* 58: 1624–1637.

Slack KV, Tilley LJ, Kennelly SS. 1991. Mesh-size effects on drift sample composition as determined with a triple-net sampler. *Hydrobiologia* 209: 215–226.

Turner D, Williams DD. 2000. Invertebrate movements within a small stream: Density dependence or compensating for drift? *International Review of Hydrobiology* 85: 141–156.

Wallace JB, Webster JR. 1996. The role of macroinvertebrate in stream ecosystem function. *Annual Review of Entomology* 41: 115–139.

Waters TF. 1964. Recolonisation of denuded stream bottom areas by drift. *Transactions of the American Fisheries Society* 93: 311–315.

Waters TF. 1965. Interpretation of invertebrate drift in streams. *Ecology* 46: 327–334.

Waters TF. 1966. Production rate, population density, and drift of a stream invertebrate. *Ecology* 47: 595–604.

Weber N, Bouwes N, Jordan C. 2017. Accounting for spatial and temporal variation in macroinvertebrate community abundances when measuring the food supply of stream salmonids. *Freshwater Science* 36: 364–376.

Williams DD, Williams NE. 1993. The upstream/downstream movement paradox of lotic invertebrates: quantitative evidence from a Welsh mountain stream. *Freshwater Biology* 30: 199–218.

Zuur AF, Leno EN, Walker NJ, Saveliev AA, Smith GM. 2009. Mixed Effects Models and Extension in Ecology with R. In: Gail M, Krickeberg K, Samet J, Tsiatis A, Wong W (Eds), *Statistics for biology and health*. Springer Science, New York. pp 1–563.

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