Estimating benthic secondary production from aquatic insect emergence in streams affected by mountaintop removal coal mining, West Virginia, USA

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Abstract:

Mountaintop removal and valley fill (MTR/VF) coal mining recountours the Appalachian landscape, buries headwater stream channels, and degrades downstream water quality. The goal of this study was to compare benthic community production estimates, based on seasonal insect emergence, between mined and forested streams in the Twentymile Creek watershed, WV (USA). We also assessed the relationship between structural and functional indicators by comparing our production estimates to traditional bioassessment measures. Emergence traps were deployed seasonally for 2-4 weeks beginning in Autumn 2007 along 100-m reaches in each of five mined and five forested streams. The study reaches in the mined streams were located at varying distances downstream of their respective valley fills. Benthic community production was calculated using published length-mass equations and emergence:production ratios. No differences in seasonal emergent density (indiv. m^{-2} d^{-1}), biomass (mg m^{-2} d^{-1}) or annual secondary production (g AFDM m^{-2} y^{-1}) were detected between treatments. Annual secondary production estimates for mined streams were highly variable and averaged 29.6 g AFDM m^{-2} y^{-1}, but ranged from 1.51 g AFDM m^{-2} y^{-1} in the stream nearest to its valley fill to 65.69 g AFDM m^{-2} y^{-1} in another stream that was 1 km downstream from its fill. Production of forested streams was more consistent with an average of 20.42 g AFDM m^{-2} y^{-1} and ranged only from 13.81-27.17 g AFDM m^{-2} y^{-1}. Annual production estimates were not correlated with benthic community index scores, component metrics, or habitat assessment scores. Only EPT production estimates of were significantly correlated with structural endpoints. Conductivity of mined streams was >30X greater than forested streams and contributed to strong differences in emergence composition. Chironomids alone accounted for >80% of production in mined streams while forested streams had significantly higher EPT production. Measures of stream ecosystem function, including secondary production, can provide a more holistic stream assessments. Prior to their widespread application as indicators of stream health, however, studies are needed to further develop robust response functions across disturbance gradients from multiple stressors.

Keywords: Appalachian, bioassessment, Chironomidae, conductivity, ecosystem function, EPT, indicator, stream assessment, valley fill

INTRODUCTION

Coal-fired power plants currently generate about half the electricity used in the United States, despite changing energy policies that promote greater use of alternative fuels. In 2007 nearly one quarter of all U.S. coal came specifically from the Central Appalachian Mountains, primarily eastern Kentucky, southern West Virginia, and Virginia (USDOE 2008a,b). Much of the coal is this region is highly valued for its relatively high heating capacity and low sulfur content. Low sulfur coal is in demand as a result of more stringent sulfur dioxide (SO_2) emissions standards established by the Federal Clean Air Act Amendment of 1990 (Fox 1999).

Coal mining in the central Appalachians has increasingly shifted from underground or contour mining to the use of large scale mountaintop removal and valley filling (MTR/VF) operations as a result of the increased demand and economic viability. By the MTR/VF method of extraction, explosives and heavy equipment are used to completely remove overlying mountaintops to access coal seams. The excess overburden, or spoil, is then placed in the adjacent valleys, creating a valley fills that permanently bury headwater stream channels. Though such fills often result from other forms of mining or development, by far the largest are associated with MTR/VF activities. From 1985 to 2001 alone, an estimated 724 stream miles were buried under fills in Appalachia (USEPA 2005).

Aside from direct loss of stream channels by burial, there are numerous indirect effects of MTR/VF on downstream resources. Sediment control ponds are constructed below valley fills to facilitate removal of particulates, though fine sediments can still be elevated downstream (Wiley et al. 2001). Movement of water through unconsolidated fill material also leeches ions and results in highly elevated total dissolved solids (measured as specific conductance) downstream (e.g., USEPA 2005, Johnson et al. 2010). Stream conductivity at filled sites can be 100X greater than for unmined streams in the region (USEPA 2005) (i.e., from ca. 50 μS cm^{-1} to 5,000 μS cm^{-1}) and evidence suggests that conductivity can remain high for decades after fill construction (Merricks et al., 2007). Other adverse effects include stabilized water temperatures, elevated base flow, increased flood potential, and impaired benthic macroinvertebrate communities (Wiley et al. 2001, USEPA 2005, Hartman et al. 2005, Pond et al. 2008, Griffith et al. 2012).

Litigation and need for functional measures

Mining coal by MTR/VF is highly controversial and the practice has been the subject of much litigation. In a 2007 federal court ruling (Ohio Valley Environmental Coalition v. USACE, Southern District, West Virginia), the need for measurement of stream ecosystem functions was highlighted because appropriate mitigation practices require replacement of functions lost by burial of headwater streams. Yet functions are not typically measured, neither in the streams prior to burial nor in resulting mitigation areas. Stream assessment methods have instead relied solely on conventional structural measures (e.g., EPA rapid bioassessment protocols [Barbour et al. 1999]) that focus primarily on species abundance, habit, feeding mode, and assemblage composition. Qualitative measures of habitat structure are also typically incorporated. These structural measures of stream health have a proven history in stream assessment and are widely applied by state agencies to report on stream condition as required by §305(b) of the Clean Water Act. Structural assessment measures, however, are only snapshots in time that may say little of underlying ecological processes. Measures of both structure and function are needed for more holistic characterization of stream health (e.g., Gessner and Chauvet 2002, Young et al. 2008, Fritz et al. 2010, Clapcott et al. 2011).

Stream ecosystem functions are typically expressed as rates and include a wide variety of measures (e.g., metabolism, nutrient uptake, organic matter decomposition, primary and secondary production) that describe energy flow and how materials are stored, processed, or transported within a system. Functional measures have been widely used in basic research to characterize ecosystem processes worldwide and to assess functional responses to various disturbance types and experimental manipulations (e.g., Carpenter et al. 2001, Peterson et al. 2001, Cross et al. 2006). Most functional measures are, however, more time, labor, or cost intensive than structural measures so they have not been widely applied by regulatory agencies for stream assessment purposes. Compared to studies using structural measures, fewer studies have measured functions in response to anthropogenic disturbance where multiple stressors may be present, and most have focused on organic matter decomposition (Guessner and Chauvet 2002, Dangles et al. 2004, Hagen et al. 2006, Fritz et al. 2010, Clapcott et al. 2011) rather than functions that can be more difficult to measure, such as stream metabolism (but see Bunn et al. 1999, Fellows et al. 2006, Young et al. 2008, Feio et al. 2010, Clapcott et al. 2011) or secondary production (but see Kedzierski and Smock 2001, Shieh et al. 2002, Carlisle and Clements 2003, Runck 2007, Woodcock and Huryn 2008).

Secondary production measures the rate of biomass accumulation in a give area through time for a given species or community (e.g., Benke and Huryn 2006). Though several different methods may be used to calculate production, all require multiple specimen collections over the entire developmental stage to measure density and biomass of the population(s) of interest (Benke and Huryn 2006). Annual production studies of temperate stream invertebrates, for example, typically involve monthly or bimonthly sampling. Suggested rapid alternatives to actual measurement of secondary production (reviewed by Benke and Huryn 2006) include application of an assumed production to biomass ratio (P:B) to measures of invertebrate biomass or use of multiple regression models that estimate production based on variables such as temperature, discharge, biomass, or maximum body size. A third rapid approach is the use of emergent aquatic insect biomass. Statzner and Resh (1993) demonstrated the significant relationship between emergent biomass and benthic secondary production for several European streams. Studies that have measured both emergence and larval production of aquatic insects have further indicated that emergence to production ratios (E:P) are relatively constant across taxonomic groups so that emergence data alone can provide accurate estimates of larval production (Poepperl 2000).

The objectives of this study were to estimate annual benthic production using seasonal emergence data and to compare insect emergence in mined and forested headwater streams. Specifically, we compared seasonal abundance, biomass, production, and composition of emergent insects. We also aimed to evaluate use of emergence as a functional assessment approach by comparing results with traditional structural measures for stream assessment.

Methods

Study Area

The Twentymile Creek watershed (ca. 225 km^2) is located in southern West Virginia, USA and drains into the Gauley River. The watershed has a trellised drainage and lies within the Dissected Plateau of the Central Appalachians [Level IV Ecoregion (69d); Woods et al. 2002]. The ecoregion is unglaciated, consisting of steep ridges and narrow valleys and is underlain with Pennsylvanian sandstone, shale, siltstone, and coal. Soils are comprised of moderately fertile Ultisols and Inceptisols (Woods et al. 1999). The watershed was clearcut in the early 1900s and some historic contour and underground coal mining has taken place in the upper portion of the watershed (Green et al. 2000). The majority of MTM/VF coal has taken place in the upper portion of the watershed in the last 20 years. Vegetation of forested subcatchments now consists primarily of mixed hardwoods, including oaks (*Quercus* spp.), maples (*Acer* spp.), tulip poplar (*Liriodendron tulipifera*) and beech (*Fagus grandifolia*), whereas mined subcatchments are primarily barren or grasslands, with patchy forest in the lower portions. The upper Twentymile watershed is uninhabited while the lower portion is low density residential. Numerous gas wells also exist throughout the watershed.

Ten subcatchments were chosen within the larger Twentymile watershed. We used a paired watershed design where five subcatchments drained nearly 100% forested land and five subcatchments drained MTR/VF operations (Table 1). Stream reaches (100 m) with perennial surface flow were chosen for study near the base of each subcatchment. Our forested reaches show legacy effects of logging and have gas wells and access roads within the catchments, but these streams are representative of the least disturbed condition within the study region. Our mined stream reaches represent a gradient of mining disturbance; two were upstream of sediment ponds (which were positioned at the base of the subcatchments) and three were at varying distances downstream of ponds (Table 1). Watershed sizes ranged from 0.59-5.27 km^2 (treatment medians: forest = 1.62 km^2; mined = 1.43 km^2).

Emergence and Production

Aquatic insect emergence was measured seasonally in each stream by deploying emergence traps for 2-4 weeks beginning Autumn 2007 (21 Sept. – 8 Oct.). Other deployment dates were 24 Jan. – 12 Feb. (Winter), 1 April – 16 April (Spring), and 14 June – 16 July (Summer), 2008. Seasonal collection dates were staggered among streams because not all could be sampled in one day, but differences in start and end dates among streams were all $\leq$ 4 d each season.

During each of the four collection periods, three emergence traps were placed in each of the 10 stream reaches, with two traps in riffle habitat and one in pool (n = 120 total traps/samples). Mean percentage riffle habitat within study reaches was 67.9% (range 44.5-80%), thus habitats were sampled roughly in proportion to their occurrence. Emergence traps were enlarged versions of Banks et al. (2007) and consisted of hinged wooden frames that, when opened, covered a rectangular area of 0.28 m^2 of streambed (Banks et al. [2007] covered 0.19 m^2). Plastic capture wells were placed between frame crosspieces and filled with preservative consisting of 2/3 glycerol and 1/3 10% buffered formalin with soapy water added to reduce surface tension. Frames were then completely enclosed within 250-µm mesh. Canvas skirts sewn to the bottom of the netting were anchored to the stream bottom with cobbles. A sheet of clear plastic was also clipped over the upper 1/3 of the trap to prevent rainwater from overfilling and diluting capture wells. Further details for trap construction are provided by Banks (2005). Upon collection, capture wells were sieved (250-µm) and preserved in 70% ethanol. Contents of each trap were preserved separately. In the laboratory, emergent aquatic insects were identified to Order or Suborder and measured to the nearest mm under a dissecting microscope (10X). Length measures were then converted to ash-free dry mass (AFDM) using published length-dry weight regression equations for adult aquatic insects (Sabo et al. 2002) with an ash conversion factor of 15% (Smock 1980).

Daily emergent density (indiv. m^{-2} d^{-1}) and biomass (mg m^{-2} d^{-1}) were calculated for each trap by dividing totals by the number of days in the incubation period. Seasonal emergent production was then calculated by multiplying daily emergence biomass by the total number of days in each season and annual emergence production was calculated as the sum of seasonal emergence production values. Annual emergent production estimates were then used to calculate annual benthic production by application of a mean emergence biomass to larval production ratio (E:P) of 18.3% (Poepperl 2000), a mean value similar to those reported in other studies that have compared stream insect emergence with benthic production (Hall et al. 1970, Speir and Anderson 1974, Jackson and Fisher 1986, Statzner and Resh 1993). Emergent density, biomass, and all production estimates were calculated for both all taxa combined and individual taxonomic groups.

Structural Assessment Measures

Benthic macroinvertebrates were collected from each stream in Spring (April) and Summer (July) 2008 using a kicknet (0.5-m-wide, 595 µm mesh). Four 0.25 m^2 kick samples were collected from riffle habitat of each stream and composited (1 m^2 total). In the laboratory, preserved samples were randomly subsampled to obtain 200 ± 20% individuals and all organisms were identified to genus level. Resulting data were used to calculate the Genus-Level Index of Most Probable Stream Status (GLIMPSS; Pond et al. 2008). This multimetric index is currently used by the West Virginia Department of Environmental Protection and the US Environmental Protection Agency (EPA) Region 3 for assessing stream health in West Virginia. Index metrics include: total generic richness; intolerant richness; Ephemeroptera generic richness; Plecoptera generic richness; clinger generic richness; Genus biotic index; % Ephemeroptera; % Orthocladiinae; and % 5 dominant genera.

We measured pH, conductivity (µS/cm), dissolved oxygen (mg/l), and in each stream during each season using a portable multiprobe (Hydrolab Quanta; Hydrolab Corp., Austin, TX). Water temperature was measured at 4-h intervals with StowAway TidbiTH temperature loggers

(OnsetH Computer Corp., Bourne, MA). Stream discharge was measured each season using the velocity area method and we used an electronic flow meter (Marsh-McBirney Flo-mate, Loveland, CO) to measure velocity.

Habitat was qualitatively scored at each site using both the US EPA Rapid Bioassessment Protocol (RBP) for high gradient streams (Barbour et al. 1999) and the Riparian, Channel, and Environmental (RCE) Inventory (Peterson 1992). The RBP has been used by regulatory agencies to assess mitigation for MTM/VF and scores each of the following variables 0-20 each for total possible score of 200: epifaunal substrate, embeddedness, velocity/depth regime, sediment deposition, channel flow status, channel alteration, frequency of riffles, bank stability, vegetative protection, and riparian vegetative zone width. The RCE scores 16 variables, each in four scoring categories that are differentially weighted, for a total possible score of 360. Total scores are then used to classify streams from 1 (excellent) to 5 (poor) with recommended management actions for each class. RCE variables are: landuse beyond riparian zone, riparian zone width, completeness of riparian zone, riparian vegetation within 10 m of channel, retention devices, channel structure, channel sediments, stream-bank structure, bank undercutting, stony substrate feel and appearance, stream bottom, riffles and pools, aquatic vegetation, fish, detritus, and macrobenthos.

Data Analyses

Preliminary results indicated no significant differences between pool and riffle traps for emergent insect abundance (paired t-test, D.F. = 28, $p = 0.62$, [two riffle traps averaged when both were recovered]) or biomass ($p = 0.52$) within streams where traps from both habitats were recovered. Pool and riffle samples were therefore averaged for each stream and collection date for calculations of emergent density, biomass, and production. Emergent insect abundance and biomass values were log(x+1) transformed and tested for differences using a generalized linear model (PROC GLM, SAS 9.1; SAS Institute, Cary, North Carolina) with season and treatment as factors and stream nested within treatment. Streams were nested within each treatment because each stream belonged to only one treatment group (e.g., mined or forested). Significant treatment effects were followed with Tukey multiple comparisons for that factor. Differences in assemblage composition between treatments were first tested using the multi-response permutational procedure (MRPP, PC-ORD v. 4.25, McCune and Mefford 1999) which has proven useful in comparing species composition data (Biondini et al. 1985; Zimmerman et al. 1985). Sensitive taxa within the Orders Ephemeroptera, Plecoptera, and Trichoptera were combined into a metric (EPT) and the combined density and biomass were tested using a generalized linear model as described above. Calculated production estimates and functional metrics (e.g. EPT production) were correlated with structural assessment measures by Pearson correlations (rP) or, if data failed to meet the assumption of normality after data transformation, Spearman correlation (r_S) (SigmaStat v. 3.0, SPSS, Inc.).

Results

Stream conductivity values ranged from 49-2513 µS/cm over the study period and the mean of mined streams (1630 µS/cm) was >30X higher than that of forested streams (52 µS/cm)(Table 2). pH of forested streams were slightly acidic to neutral (range: 5.87-6.94), whereas mined streams were more alkaline (range: 7.04-8.89). Mined streams were also, on average, ca. 1ºC warmer than forested streams over the study period. Stream discharge and dissolved oxygen were similar between treatment groups (Table 2). The average macroinvertebrate index score (GLIMPSS) for forested streams was more than double that of mined streams and habitat assessment scores (RBP and RCE) were also consistently higher in forested streams, indicating impairment of mined streams (Table 2).

Of the 120 traps deployed during the study, 92 were successfully recovered (77%) (Table 3). Traps were sometimes knocked down, either by animals or high stream flows. Trap losses were highest in summer (47%) and lowest in winter (7%), and trap losses were similar between mined and forested streams. There were no traps recovered for one forested stream in Autumn (Peters) and three streams in Summer (one forested [Ash], two mined [Buckles and Lost]).

There were significant differences in insect emergence among seasons for both density and biomass (Table 4), with greatest emergence in summer and lowest in winter (Figure 1). There was, however, no significant treatment effect for either density or biomass of emergent insects over the study period. There were differences among streams within treatments (stream[treatment]) that resulted largely from differences between mined streams with study reaches located upstream of sediment ponds versus those with study reaches located downstream of sediment ponds (Figure 1, Table 4).

Emergence was often highly variable among streams. In winter, emergent density and biomass ranged from only 0.90 indiv. m^{-2} d^{-1} and 0.26 mg m^{-2} d^{-1}, respectively, in Lost Creek (mined, upstream of sediment pond and closest to a valley fill) to 17.41 indiv. m^{-2} d^{-1} and 9.34 mg m^{-2} d^{-1} in Buckles Branch (mined, downstream of sediment pond). Winter densities averaged only 2.81 and 5.21 indiv. m^{-2} d^{-1} for forested and mined streams, respectively (Fig. 1) . Winter biomass averaged 1.26 mg m^{-2} d^{-1} in forested streams and 2.81 mg m^{-2} d^{-1} in mined streams. During peak summer emergence, no traps were recovered in either Lost Creek or Buckles Branch, but emergence rates were lowest in Beech Fork (11.53 indiv. m^{-2} d^{-1}, 1.98 mg m^{-2} d^{-1}), the other mined site upstream of a sediment pond, and highest in Sugarcamp Creek (171.21

indiv. m^{-2} d^{-1}, 91.10 mg m^{-2} d^{-1}), which like Buckles, is a mined site located >1 km downstream of the valley fill and sediment pond. Summer density and biomass averaged 91.95 indiv. m^{-2} d^{-1} and 22.93 mg m^{-2} d^{-1}, respectively in forested streams, whereas mined streams averaged 75.77 indiv. m^{-2} d^{-1} and 34.44 mg m^{-2} d^{-1} (Fig. 1). Emergence rates from forested and mined streams in both autumn and spring were intermediate to those in winter and summer. The lowest emergence rates in autumn and spring were from Lost Creek, where emergence rates (Autumn: density = 0.97 indiv. m^{-2} d^{-1}, biomass = 0.34 mg m^{-2} d^{-1}; Spring: density = 1.40 indiv. m^{-2} d^{-1}, biomass = 0.44 mg m^{-2} d^{-1}) were >20X lower than the season average of all streams combined.

For annual benthic production calculations, when no traps were recovered in a stream (4 occasions) we used the treatment average for that season. However, for one mined stream in Summer (Lost) we used the value of the other mined site upstream of a sediment pond (Beech) because of differences in emergence at sites upstream and downstream of sediment ponds. Conversion of seasonal emergent biomass to benthic production resulted in annual averages of 20.42 g AFDM m^{-2} yr^{-1} for forested streams and 29.60 g AFDM m^{-2} yr^{-1} for mined streams. Secondary production from mined streams showed much greater variability, however, ranging from 1.51 g AFDM m^{-2} yr^{-1} (Lost Creek) to 65.69 g AFDM m^{-2} yr^{-1} (Buckles Branch), whereas forested streams ranged just 13.81 g AFDM m^{-2} yr^{-1} (Peters Creek) to 27.17 g m^{-2} yr^{-1} (Jacks Branch; Fig. 2).

While there was no treatment effect on emergent insect abundance, biomass, or production, there were significant differences in taxonomic composition. Chironomids dominated emergence in all seasons and generally comprised at least half of all insect production in each stream (Fig. 2, Appendix). Forested streams had greater production attributed to EPT taxa, while production in mined streams was dominated by Chironomidae. Differences in composition between forested and mined streams were significant during both Spring (MRPP, A = 0.09, p = 0.021) and Summer (A = 0.12, p = 0.037) emergence periods. There were no significant composition differences in either Autumn (A = 0.03, p = 0.185) or Winter (A = 0.008, p = 0.39) when emergent samples consisted largely of Chironomidae and winter stoneflies. There was also a highly significant treatment effect when only emergent density and biomass of EPT taxa were considered (Table 5). In Spring, EPT density and biomass of forested streams averaged 1.97 indiv. m^{-2} d^{-1} and 5.06 mg m^{-2} d^{-1}, respectively, whereas mined streams averaged only 0.16 indiv. m^{-2} $d^{-1}$and 0.88 mg m^{-2} d^{-1}. Differences were even greater in Summer when forested streams averaged 9.62 indiv. m^{-2} d^{-1} and 11.35 mg m^{-2} d^{-1}, and mined streams averaged 0.96 indiv. m^{-2} $d^{-1}$and 1.00 mg m^{-2} d^{-1}. Though two mined streams, Sugarcamp and Buckles, had the highest annual production measured in this study (52.27 g AFDM m^{-2} yr^{-1} and 65.69 g AFDM m^{-2} yr^{-1}, respectively), Nematocera (primarily Chironomidae) alone accounted for ca. 87% of benthic production (45 and 57.8 g AFDM m^{-2} yr^{-1}, respectively) in each of these streams (Fig. 2). In the forested streams chironomid production never accounted for more than 54% (in Neil) of total insect production, while EPT taxa accounted for, on average, 48% of production (Fig. 2, Appendix). EPT production in mined streams remained consistently low across seasons, the average ranging from only 0.30 g AFDM m^{-2} yr^{-1} (Winter) to 0.48 g AFDM m^{-2} yr^{-1} (Summer), and was comprised primarily of nemourid stoneflies and hydropsychid caddisflies. Mayflies (Ephemeroptera) were conspicuously rare in mined streams where average production was only 0.13 g AFDM m^{-2} yr^{-1} (0.4% of total), compared to 4.4 g AFDM m^{-2} yr^{-1} (22% of total) in forested streams.

Annual secondary production estimates for each stream were not correlated with any of the 9 physiochemical and structural assessment measures included in this study (from Table 2, Pearson correlation coefficients range: -0.29 to 0.48; $p > 0.05$). Annual production of combined EPT taxa, however, was significantly correlated with 7 of the 9 variables: conductivity (rS = -.939, p = <0.001), pH (rP = -0.785, $p = 0.007$), temperature (rP = -0.744, $p = 0.014$), degree days (rP =-0.798, $p = 0.006$), GLIMPSS (rP = 0.815, $p = 0.004$), RBP (rP = 0.768, $p = 0.010$), and RCE (rP = 0.769, $p = 0.009$).

DISCUSSION

Our findings provide an example of how structural and functional measures can provide unique insights to stream ecosystem health. We found strong structural differences (e.g., species composition) between mined and forested streams, but secondary production estimates were highly variable and were not, on average, different between treatments. Variability of mined streams appeared to be related to distance from valley fills, with production generally increasing with distance from valley fills. Total benthic community production estimates correlated poorly with structural assessment parameters. Only production of sensitive EPT taxa correlated well with the commonly used structural measures. The relative absence of sensitive taxa, particularly Ephemeroptera, and high TDS we observed in mined streams supports previous studies in Appalachian streams affected by coal mining activities (Hartman et al. 2005, Merricks et al. 2007, Pond et al. 2008, Pond 2010, Griffith et al. 2012). Further mechanistic studies (sensu Buchwalter and Luoma 2005, Xie et al. 2010) of ecophysiology are needed to fully explain sensitive species losses in these mined streams.

Lack of overall difference in emergence abundance, biomass, and production between treatments results from prevalence of pollution tolerant chironomids in mined streams that was offset with a corresponding reduction of sensitive EPT taxa. When insect emergence rates were averaged by treatment, we found no differences within any season. We found expected seasonal peak emergence in summer and low in winter, however the similarity of emergence rates in autumn and spring was surprising. Autumn emergence, however, consisted almost entirely of chironomids, and had more, smaller individuals, whereas Spring samples consisted of fewer, larger individuals, and more EPT taxa (Appendix). Summer samples, particularly among mined streams, had high variability resulting largely from trap losses during that period.

Our mean benthic community production estimate of 25 g m^{-2} yr^{-1} across all streams would be considered fairly typical of lotic systems (e.g., Huryn and Wallace 2000), though production in mined streams was highly variable and appeared to be related to distance from the valley fill. The two mine sites upstream of the sediment ponds (Lost and Beech) had the lowest secondary production measured here, whereas the two mined sites furthest downstream of fills (Buckles and Sugarcamp) had significant higher production than reference streams. Our calculated annual production of 1.5 g AFDM m^{-2} yr^{-1} for mined Lost Creek is among the lowest reported for streams (Huryn and Wallace 2000). The sample reach in Lost was upstream of the sediment pond and had the lowest habitat score (e.g., RBP = 119) of any stream in this study. This study site was also closest to the valley fill, had highly unstable channel with substrates consisting of gravel, sand, and fines that had washed down from the fill. Riparian vegetation was also largely lacking and the stream reach would be considered highly impaired. Beech Creek, the other mined stream upstream of the sediment pond had more a more typical production value of 10.8 g AFDM m^{-2} yr^{-1}, though still second lowest of the streams measured and comprised primarily of Chironomidae. Unlike Lost Creek, Beech Creek has an intact riparian zone and larger, more stable substrates that improved habitat quality relative to Lost. In many cases, a single sediment pond will serve multiple valley fills and the length of stream between the toe of fills and sediment ponds can extend hundreds of meters.

Contrasting the low production of Lost and Beech Creeks, our estimated benthic production in two mined streams located furthest from valley fills had much higher production than any other stream, despite no obvious increase in basal resources, such as periphyton or particulate organic matter (unpublished data). Higher secondary production in Sugarcamp and Buckles, was largely attributed to chironomid production values of 45 and 58 g AFDM m^{-2} yr^{-1}, respectively, which comprised 87% of total benthic production in each stream. These estimated values are similar to the high chironomid production values reported for Sycamore Creek, AZ (Jackson and Fisher 1986) and snag habitats of the Ogeechee River, GA (Benke 1998), which may represent extreme examples of high midge production for undisturbed streams. Furthermore, our estimates are very similar to those of Runck (2007) who calculated chironomid production of 59.5 g AFDM m^{-2} in an industrially contaminated, nutrient-enriched stream. Similarly, Shieh et al. (2002) found higher production of non-tanypodine chironomids (59.2 g AFDM m^{-2} y^{-1}) in a stream receiving treated wastewater and attributed excess production to increased nutrients and reduced predation pressure. In these mined streams, high chironomid production could also have resulted from their tolerance to degraded water quality or from indirect effects such reduced competition and predation pressure. Temperature significantly influences macroinvertebrate growth and production (e.g., Huryn and Wallace 2000) and Sugarcamp creek averaged ca. 1.5 º warmer than other streams and accumulated 430 more DD. However, the stream with the highest production, Buckles Creek, had accumulated lowest degree days among the mined streams (Table 2). Predators may be less abundant as a result of degraded water quality (e.g., high TDS) or lack their preferred prey. When prey resources shift from larger bodied taxa (e.g., EPTs) to smaller-bodied midges, predators would theoretically have to expend more energy searching for a greater number of smaller individuals. More food web studies are needed, however, that document food web energetics of disturbed streams (i.e., Runck 2007).

The relationship between ecosystem structure and functions has received much attention (e.g., Loreau et al. 2001, Fritz et al. 2010, Isbell et al. 2011). Odum (1985) suggested that ecosystem functions may be less responsive to stress than structural measures such as species diversity, and this has been supported by several researchers (e.g. Schindler 1985, Frost et al. 1995, Niyogi et al. 2002). This functional complementarity, or redundancy, can occur when disturbance causes shifts in species composition, yet ecosystem functions remain largely unaffected (e.g., Frost et al. 1995), suggesting an inherent stability of process. We found that traditional stream assessment measures, such as EPT metrics and habitat scores, were consistently responsive to mining disturbance in our study streams, only when considering production of combined EPT taxa did we find correlations with structural indicators. Measures of community production were, however, highly variable among mined sites, whereas values from forested streams were relatively consistent. Though we found no differences in *average* production between treatments, our findings highlight the unique benefits of functional assessment methods. Variability of production estimates at mined sites are likely related to not only water quality and habitat, as structural indicators typically reflect, but also the underlying resource base, food web structure, and system energetics.

Studies of anthropogenic disturbance effects on stream ecosystem functions are increasing, but their sensitivity to disturbance may depend on the specific function chosen, stressor(s) present, and system in question. This study was limited to secondary production in a small number of mined streams and further research is needed to fully evaluate utility of functional assessment methods. Measures of organic matter decomposition are among the most straightforward functions to measure and thus far have shown most promise as an indicator of stream health (reviewed in Gessner and Chauvet 2002). Additional studies are needed that further assess strength and direction of functional responses, sensitivity to various disturbance types, and the accuracy and precision measurements across larger spatial scales. Use of functional measures is not intended as a replacement for existing methods used to assess stream health. Rather, they are intended as a complement to provide process information that will aid regulatory and stream mitigation programs to prevent loss of ecosystem functions. Further, no single functional assessment method will properly evaluate ecosystem processes. A suite of functional measures are recommended to provide a more thorough, holistic assessment of stream ecosystem health (Young et al. 2008, Feio et al. 2010, Clapcott et al. 2011).

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Table 1. Watershed and stream channel characteristics for ten study streams in the Twentymile Creek watershed, WV. Categories are Forested (F), Mined Upstream of ponds (MU), and Mined downstream of ponds (MD). Upstream channel length represents total of main channel and all tributaries upstream based on National Hydrologic Database 1:24,000 scale maps.

			Major Landuses (%)		**Channel Substrates (%)**			
Stream	**Category**	**Watershed Area (km^2)**	**Deciduous Forest**	**Barren**	**Mixed Substrates**	**Bedrock Outcrop**	**Upstream Channel Length (km)**	**Distance to Valley Fill (km)**
Ash	F	3.99	100.0	0	95	5	5.89	—
Jack	F	0.91	99.7	0	80	20	1.26	—
Laurel	F	1.41	100.0	0	96	4	1.56	—
Neil	F	3.73	99.8	0.1	99	1	5.46	—
Peters	F	1.62	94.6	1.2	99	1	2.79	—
Lost	MU	0.79	8.1	91.9	97	3	1.06	0.16
Beech	MU	0.57	36.1	63.9	73	27	0.75	0.33
Hardway	MD	1.43	40.6	59.4	68	32	2.58	0.53
Sugarcamp	MD	5.27	18.9	77.4	97	3	8.73	1.40
Buckles	MD	2.61	61.9	38.1	98	2	3.71	0.97

Table 2. Stream discharge and physicochemistry for study streams in the Twentymile Creek Watershed, WV. Categories are Forested (F), Mined Upstream of ponds (MU), and Minded Downstream of ponds (MD). Temperature values are daily mean, others are mean of four seasonal measures (±S.E). Temperature and degree days cover entire study period (27 Sept. 07 to 14 July 08). GLIMPSS = Genus-level Index of Most Probable Stream Status from macroinvertebrate sampling, RBP = Rapid Bioassessment Protocol habitat score, and RCE = Riparian, Channel, and Environmental Inventory habitat score.

Stream	**Category**	**Discharge (m^3/s)**	**Conductivity (µS/cm)**	**pH**	**D.O. (mg/l)**	**Temp. (°C)**	**Accumulated Degree Days**
Ash	F	0.076 (0.04)	41.0 (4.5)	6.20 (0.25)	9.93 (1.09)	9.88 (0.28)	2866
Jack	F	0.013 (0.01)	45.6 (1.2)	6.71 (0.16)	9.66 (1.51)	10.26 (0.27)	2980
Laurel	F	0.030 (0.02)	41.6 (4.5)	6.17 (0.19)	10.00 (1.50)	9.75 (0.26)	2827
Neil	F	0.073 (0.04)	49.3 (4.7)	6.35 (0.25)	10.41 (1.31)	10.49 (0.27)	2962
Peters	F	0.021 (0.01)	81.9 (4.5)	6.52 (0.13)	10.06 (1.17)	10.17 (0.28)	3051
Forested Average		**0.043 (0.01)**	**51.9 (7.7)**	**6.39 (0.10)**	**10.01 (0.12)**	**10.11 (0.12)**	**2937**
Lost	MU	0.012 (0.002)	2479.5 (45.9)	7.73 (0.15)	10.60 (0.85)	11.30 (0.12)	3297
Beech	MU	0.008 (0.001)	1342.1 (99.3)	7.17 (0.10)	10.63 (1.52)	10.66 (0.24)	3105
Hardway	MD	0.024 (0.01)	1706.7 (222.3)	7.50 (0.01)	10.25 (1.08)	10.93 (0.31)	3177
Sugarcamp	MD	0.073 (0.01)	1452.6 (251.6)	8.24 (0.29)	9.05 (1.26)	11.91 (0.37)	3469
Buckles	MD	0.048 (0.01)	1170.9 (74.6)	7.94 (0.07)	10.75 (0.99)	10.48 (0.23)	3063
Mined Average		**0.033 (0.01)**	**1630.4 (229.4)**	**7.72 (0.18)**	**10.26 (0.31)**	**11.06 (0.12)**	**3222**

Table 2 (cont.)

GLIMPSS	RBP Total	RCE Total
86.02	164	321
80.08	170	326
86.29	176	320
88.87	163	325
85.16	167	326
85.28	**168.0**	**323.6**
19.49	119	172
42.20	147	291
57.72	136	240
22.04	155	253
34.95	154	281
35.28	**142.2**	**247.4**

Table 3. Emergent trap recovery by stream, season, and habitat in headwater tributaries within the Twentymile Creek, WV watershed, Autumn 2007-Spring 2008. Categories F, MU, and MD as defined in Table 1.

		Autumn		**Winter**		**Spring**		**Summer**	
		Riffle	Pool	Riffle	Pool	Riffle	Pool	Riffle	Pool
	Category								
Ash	F	1	0	2	1	2	1	0	0
Jack	F	2	1	2	1	2	1	2	1
Laurel	F	2	1	2	1	2	1	1	0
Neil	F	1	1	1	1	2	1	2	1
Peters	F	0	0	2	1	2	0	1	1
Forested Totals		**6**	**3**	**9**	**5**	**10**	**4**	**6**	**3**
Lost	MU	2	0	2	1	2	1	0	0
Beech	MU	1	0	2	1	2	1	0	1
Hardway	MD	1	0	2	1	2	1	2	1
Sugarcamp	MD	2	1	1	1	2	1	2	1
Buckles	MD	2	1	2	1	2	1	0	0
Mined Totals		**8**	**2**	**9**	**5**	**10**	**5**	**4**	**3**

Table 4. Statistical results for comparison of emergent aquatic insect density and biomass by treatment (mined versus forest) and season (autumn, winter, spring and summer), * indicates significant differences ($p<0.05$).

Dependent Variable	Source of Variation	d.f.	SS	MS	F	*p*-value
Total Density	Full Model	15	11.80	0.79	4.73	0.0008*
	Treatment	1	0.32	0.32	1.91	0.182
	Stream(Treatment)	8	4.84	0.60	3.64	0.0091*
	Season	3	5.62	1.87	11.27	0.0002*
	Season*Treatment	3	0.36	0.12	0.73	0.55
Total Biomass	Full Model	15	10.74	0.72	3.91	0.0026*
	Treatment	1	0.12	0.12	0.63	0.4374
	Stream(Treatment)	8	6.23	0.78	4.25	0.0041*
	Season	3	3.69	1.23	6.71	0.0026*
	Season*Treatment	3	0.21	0.07	0.38	0.77

Table 5. Statistical results for comparison of emergent density and biomass of combined Ephemeroptera, Plecoptera, and Trichoptera (EPT) by treatment and season. * indicates significant differences (p<0.05).

Dependent Variable	Source of Variation	d.f.	SS	MS	F	*p*-value
Total EPT Density	Full Model	15	5.23	0.35	6.52	<0.0001*
	Treatment	1	1.52	1.52	28.45	<0.0001*
	Stream(Treatment)	8	1.31	0.16	3.06	0.0202*
	Season	3	1.34	0.45	8.37	0.0008*
	Season*Treatment	3	0.33	0.11	2.07	0.137
Total EPT Biomass	Full Model	15	6.67	0.44	5.47	0.0003*
	Treatment	1	2.34	2.34	28.76	<0.0001*
	Stream(Treatment)	8	1.74	0.22	2.68	0.0354*
	Season	3	1.49	0.49	6.09	0.0041*
	Season*Treatment	3	0.49	0.16	2.02	0.1432

Appendix. Emergent density, Dens (individuals m^{-2} d^{-1}), biomass, Biom (mg m^{-2} d^{-1}), and estimated benthic production, Prod (mg AFDM m^{-2} season/year^{-1}). Seasonal values are average of streams within treatment. Annual values for A and B are averages across seasons, whereas annual benthic P represent sum of seasonal values. Significant differences between treatments within A, B, or P categories for individual taxa/metric in bold.

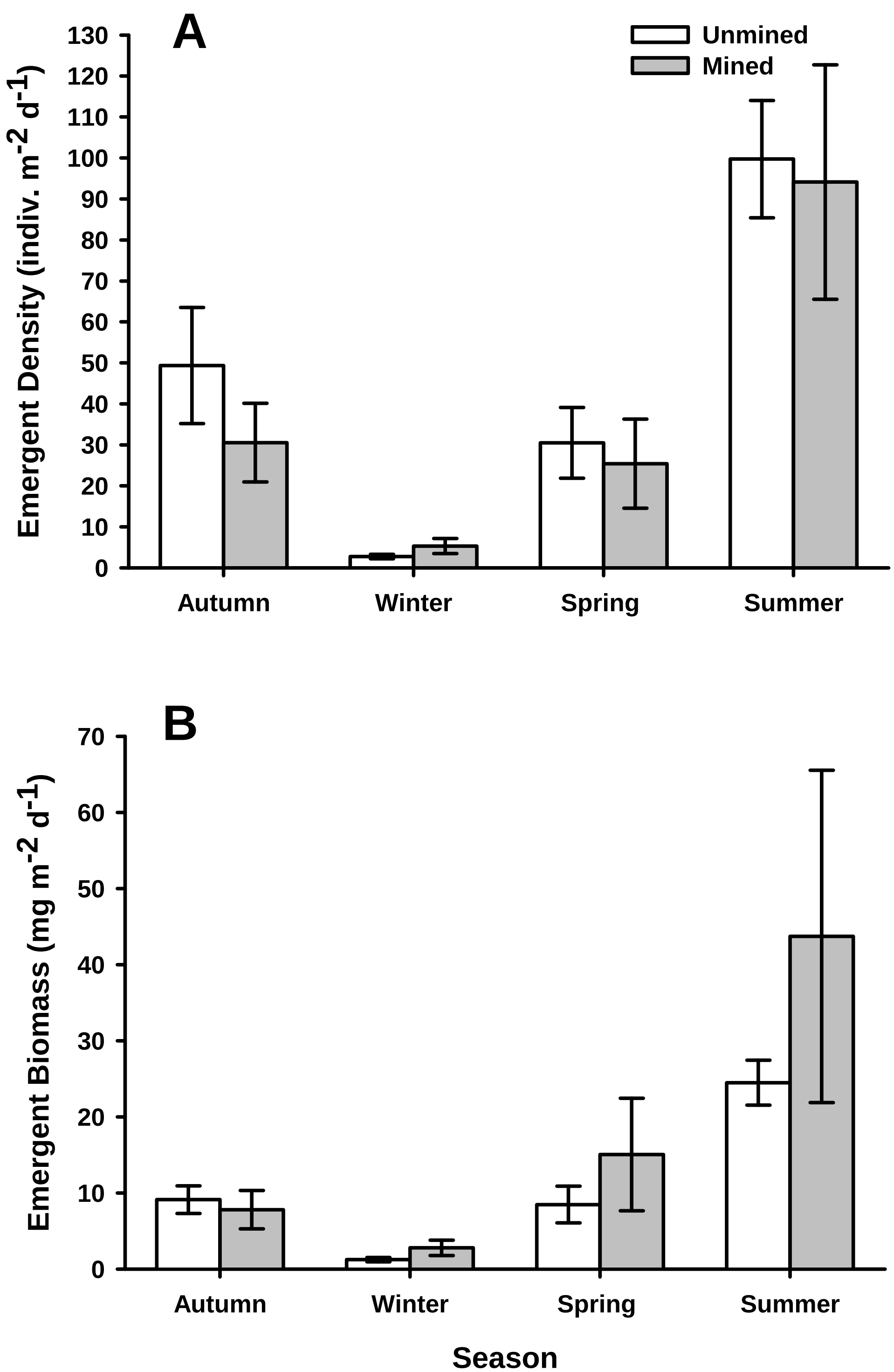
Taxon/Metric	Treatment	Season/Year	Dens	Biom	Prod
Nematocera	Mined	Autumn	35.53	9.28	4613.3
(Diptera)		Winter	4.44	2.21	1084.6
		Spring	25.17	14.08	7079.1
		Summer	72.52	13.20	6634.2
		Annual	*34.41*	*9.69*	*19411.2*
	Forest	Autumn	42.51	4.92	2445.2
		Winter	1.90	0.43	209.2
		Spring	27.35	3.05	1535.5
		Summer	81.97	11.54	5801.4
		Annual	*38.43*	*4.98*	*9991.3*
Brachycera	Mined	Autumn	0.29	0.06	30.8
(Diptera)		Winter	0.01	0	0.3
		Spring	0.07	0.09	44.8
		Summer	2.23	0.22	111.1
		Annual	*0.65*	*0.09*	*187.0*
	Forest	Autumn	0.73	0.07	36.0
		Winter	0.05	0	1.8
		Spring	0.19	0.04	18.6
		Summer	0.36	0.04	18.2

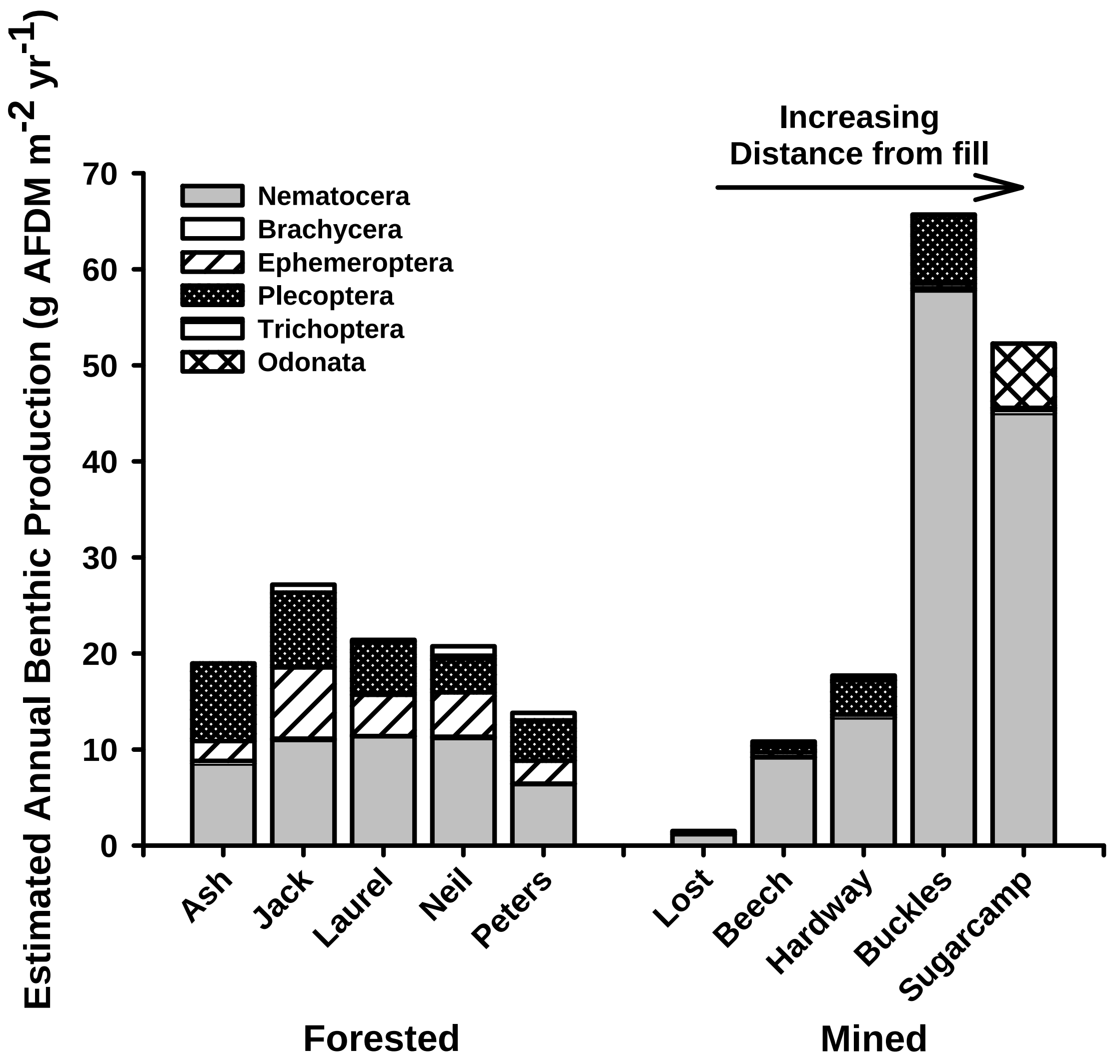
		Annual	*0.33*	*0.04*	*74.6*
Ephemeroptera	Mined	Autumn	0.02	0.04	18.0
		Winter	0	0	0
		Spring	0.04	0.23	116.1
		Summer	0	0	0
		Annual	*0.02*	*0.07*	*134.1*
	Forest	Autumn	0.76	0.99	492.3
		Winter	0	0	0
		Spring	1.65	3.85	1933.4
		Summer	4.82	3.93	1977.9
		Annual	*1.81*	*2.19*	*4403.7*
Plecoptera	Mined	Autumn	0.08	0.41	202.5
		Winter	0.62	0.60	295.5
		Spring	0.10	0.62	309.7
		Summer	0.05	0.17	83.7
		Annual	*0.21*	*0.45*	*891.4*
	Forest	Autumn	0.43	2.28	1133.0
		Winter	0.85	0.83	407.5
		Spring	0.09	0.95	479.9
		Summer	1.67	5.19	2610.4
		Annual	*0.76*	*2.31*	*4630.9*
Trichoptera	Mined	Autumn	0.47	0.21	106.4
		Winter	0	0	0

		Spring	0.02	0.03	15.4
		Summer	0.91	0.84	420.7
		Annual	*0.35*	*0.27*	*542.5*
	Forest	Autumn	0.12	0.14	71.8
		Winter	0	0	0
		Spring	0.22	0.26	131.7
		Summer	3.13	2.22	1119.7
		Annual	*0.87*	*0.66*	*1323.3*
Odonata	Mined	Autumn	0	0	0
		Winter	0	0	0
		Spring	0	0	0
		Summer	0.06	20.02	10064.1
		Annual	*0*	*0*	*10064.1*
	Forest	Autumn	0	0	0
		Winter	0	0	0
		Spring	0	0	0
		Summer	0	0	0
		Annual	*0.01*	*5.00*	*0*
Total EPT	Mined	Autumn	0.57	0.66	327.0
		Winter	0.62	0.6	295.5
		Spring	0.16	0.88	441.2
		Summer	0.96	1.00	504.3
		Annual	*0.58*	*0.78*	*1567.9*

Forest	Autumn	1.31	3.41	1697.2
	Winter	0.85	0.83	407.5
	Spring	1.97	5.06	2545.1
	Summer	9.62	11.35	5708.1
	Annual	*3.44*	*5.16*	*10357.9*

List of Figures


A
Unmined
Mined
Emergent Density (indiv. m-2 d-1)
130
120
110
100
90
80
70
60
50
40
30
20
10
0
Autumn
Winter
Spring
Summer
B
Emergent Biomass (mg m-2 d-1)
70
60
50
40
30
20
10
0
Autumn
Winter
Spring
Summer
Season


Estimated Annual Benthic Production (g AFDM m-2 yr-1)
Increasing
Distance from fill
Nematocera
Brachycera
Ephemeroptera
Plecoptera
Trichoptera
Odonata
0
10
20
30
40
50
60
70
Ash
Jack
Laurel
Neil
Peters
Lost
Beech
Hardway
Buckles
Sugarcamp
Forested
Mined