

Supplement A: Creating an Ecopath Model for Headwater Streams in the Salmon River Basin

PARAMETER ESTIMATES

Periphyton.—Periphyton standing stocks were estimated from mid-summer samples (August 4, 2004). Ten rocks were sampled at each site on each date. The standing stock biomass estimates (in g of ash-free dry mass [AFDM]/m²) were corrected for estimated embeddedness of the rocks. Rocks in the South Fork of the Salmon River (SFS) were 33% embedded. Data on net primary production were not available for the specific stream evaluated here. Therefore, in accordance with parameter borrowing procedures for Ecopath with Ecosim (EwE; Dame and Christian 2008; Coll et al. 2009), data from comparable streams were applied. Minshall et al. (1983, 1992) conducted extensive stream surveys in the Snake River basin during the 1980s and 1990s, and they estimated gross primary production and total ecosystem metabolism for a range of streams. In those studies, Smiley and Obsidian creeks were the most comparable in size to SFS. Minshall et al. (1992) estimated gross primary productivity to be 0.38 g of carbon·m⁻²·d⁻¹ in Smiley Creek and 0.49 g of carbon·m⁻²·d⁻¹ in Obsidian Creek. The total community respiration at these sites was -0.26 and -0.40 g of carbon·m⁻²·d⁻¹, respectively, so the minimum net primary productivity (NPP) would be between 0.09 and 0.12 g of carbon·m⁻²·d⁻¹. However, the total metabolism takes into account both autotrophic and heterotrophic respiration over a 24-h period; Minshall et al. (1992) specifically noted that they evaluated the respiration of detritus separately and included that in the reported total daily respiration estimate. Therefore, actual NPP is assuredly larger than the minimum values noted above and reported as net daily metabolism by Minshall et al. (1992). If we assume that one-half of the net daily metabolism is derived from heterotrophs and one-half is derived from autotrophs, then the estimated NPP is 0.26 g of carbon·m⁻²·d⁻¹ for Smiley Creek and 0.28 g of carbon·m⁻²·d⁻¹ for Obsidian Creek. It is generally assumed that periphyton and other autotrophs are about 50% carbon. We therefore multiplied these production values by 2 and used the mean as the estimated daily NPP for periphyton in our study streams. We then assumed a 200-d growing season to convert this to an annual production estimate for the model input.

Macrophytes.—Estimates of macrophyte standing stock biomass (g AFDM/m²) in each stream were determined from empirical data that were collected during mid-summer stream surveys; however, empirical estimates of macrophyte production were not available for our specific study sites. We used published data on stream macrophyte growth to estimate production rates. Few studies have quantified macrophyte growth in streams. Riis et al. (2009) studied growth rates of four common stream macrophytes in rivers in Denmark (*Elodea canadensis*, *Myriophyllum spicatum*, *Potamogeton perfoliatus*, and *Ranunculus baudotii* × *pseudofluitans*); those authors found that the relative growth rates for *E. canadensis* and *M. spicatum* were between 0.06 and 0.09 per day, whereas relative growth rates for *Ranunculus* spp. and *P. perfoliatus* were about half those rates (0.02–0.04 per day). Using the mean of the ranges reported by Riis et al. (2009), we estimated the July growth rate of macrophytes in our study streams to be 0.055 per day. This is likely to be an overestimate given the differences in stream temperature and species composition of macrophytes in Snake River streams; however, with the low biomass and limited presence of macrophytes in the diets of biota in SFS, the overestimate was not expected to substantially affect the model.

Benthic organic matter (detritus).—We did not collect data on benthic organic matter (BOM) standing stocks during the summer 2004 surveys. However, Minshall et al. (1983, 1992) reported BOM estimates for comparable streams within the Salmon River basin. These values were applied to the current model. The estimated BOM standing stock used in our model (82.76 g AFDM/m²) is the mean of BOM from second- and third-order streams as reported in Table 7 of Minshall et al. (1983). Litter input into these streams ranged from 51 to 413 g·m⁻²·year⁻¹ according to Minshall et al. (1992: their Table 2). This did not account for other sources of allochthonous input, so we used a detrital import value on the higher end of this scale (375 g·m⁻²·year⁻¹).

Microbes (bacteria and fungi).—No data were available on microbial communities in our study streams. Meyer and Poepperl (2004), who created an Ecopath model for a headwater stream in Germany, estimated that bacterial biomass and fungal biomass were approximately 8% of BOM, with an overall production : biomass ratio (P/B) of 15.66 and a production : consumption ratio (P/Q) of 0.24. Given our estimated detritus standing stocks, application of the ratios from Meyer and Poepperl (2004) yielded a biomass estimate of 8.96 g/m² and a production

estimate of $140.36 \text{ g}\cdot\text{m}^{-2}\cdot\text{year}^{-1}$. We estimated the consumption rate (Q) to be $1.61 \text{ g}\cdot\text{m}^{-2}\cdot\text{year}^{-1}$ (Table 1).

Macroinvertebrates.—Three replicate Hess samples were collected at three sites on each stream during mid-summer 2004 to quantify stream macroinvertebrate biomass in the benthos. The total number of macroinvertebrates and the total biomass of the sample were quantified for each Hess sample. The larger Hess collections were subsampled for taxonomic identification. Within each subsample, all individuals were identified to the lowest possible taxonomic level (generally genus or species for aquatic insects; order or family for noninsect taxa and terrestrial taxa [of terrestrial origin, including adult forms of aquatic macroinvertebrates]). For the purposes of the Ecopath model, the aquatic invertebrate community was condensed into seven major taxonomic groups: Ephemeroptera, Plecoptera, Trichoptera, Chironomidae, other Diptera (excluding Chironomidae), Coleoptera, and noninsect aquatics. Total invertebrate AFDM was measured for each Hess sample (three samples per stream per sample date). A subset of individuals was identified to the lowest possible taxonomic level for each Hess sample. These individuals were not measured or weighed, so the estimated biomass of each larger taxonomic group could not be calculated directly. Fortunately, invertebrates were identified and measured (length, mm) from drift net samples that were collected at each site during the same sampling period as the Hess samples. For all but the noninsect aquatics, we used established length–weight relationships to estimate the mean mass of a given species or genus of invertebrates from the drift net samples (Benke et al. 1999; Baumgartner and Rothhaupt 2003). We then estimated the total biomass present for each species or genus in the Hess sample by multiplying the mean mass of that taxon (as determined from the drift net data) by the estimated number of that taxon in the Hess sample. If a species or genus in the Hess sample was not also present in the drift sample, then the mean weight of all species or genera in that order was used. The total estimated biomass for each taxonomic group was then estimated by summing the total biomass for each species or genus. The total sample biomass (calculated by adding the total biomass from each taxonomic group) was then compared with the actual measured AFDM of invertebrates. The summed biomass was consistently lower than the measured biomass in all cases. This is likely attributable to (1) the absence of larger individuals in the drift net samples due to selective predation by fish in these streams or (2) the absence of large but uncommon individuals in the subsample. We corrected for the underestimate by using the biomass estimated from the calculated values to

determine the proportional representation of each taxonomic group. These biomass proportions were then applied to the total biomass that was measured in each sample. The biomass of noninsect aquatics was assumed to match their relative abundance on a 1:1 ratio (this is the best available estimate; see below). Three Hess samples were collected at each stream during each sample date, and the values initially used for each taxonomic group in the Ecopath model represent the mean of these three samples.

Invertebrate productivity was estimated from mean biomass values, taxonomic abundance and growth rate estimates based on species or genus size (from the drift net samples as noted above), and mean stream temperature in July and August, when samples were collected. We used established growth rate equations for each order, but because species or genera varied in mean biomass, the growth rates and the estimated productivity of the taxa within orders were calculated separately for each species or genus. The estimated growth rate for each species or genus was then multiplied by the mean mass of that species or genus (as calculated from the drift net data) to estimate productivity of single individuals in that taxon. This productivity estimate was multiplied by the estimated number of individuals of that species or genus per square meter from the Hess sample, yielding an estimate of production (in $\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$) for each taxon. For each group, we then summed the production of all species or genera within that group. As with biomass, mean production for each taxonomic group was estimated from the three samples collected at each site on each date. Drift net samples of invertebrates often miss larger individuals that are preferentially selected for by fish feeding from the drift (Keeley and Grant 1997). This may lead to a slightly smaller mean invertebrate size in the drift net samples, and because smaller individuals grow faster, it may also lead to slight overestimation of the growth rate. It is therefore possible that productivity was slightly overestimated by these methods due to the smaller mean mass of invertebrates used in the growth rate calculations. However, this would be compensated for to some degree by multiplying the growth rate by this smaller biomass in estimating productivity. The estimate is also based on the standing stock of individuals, which does not account for those eaten by fish. In the absence of specific growth measurements or fish exclusion in the stream, however, we feel that our productivity estimates are reasonable and serve as the best possible estimates within the constraints of available information.

Unlike the other invertebrate taxa, individuals in the noninsect aquatic category from Hess or drift net samples were not described by head capsule or length estimates, and identification of these individuals was often limited to the order level. The number of individuals in this broad group was enumerated for each sample; it was then possible to calculate relative abundance of noninsect aquatics within the larger sample, and in the absence of size data the biomass of noninsect aquatics within the sample was estimated by simply assuming that biomass matched relative abundance on a 1:1 basis. In the absence of other data, this method likely provided a reasonable estimate of biomass for noninsect aquatics. Mean mass of individual noninsect aquatics for each sample was estimated by dividing the estimated total mass of noninsect aquatics by the number of noninsect aquatics in each sample. The estimated mass was then used with a broadly generalized growth equation to estimate growth rate and subsequent productivity of noninsect aquatics. Again, we recognize that error likely occurs in the values used in our model, but these are the best estimates given the available information.

The diet consumed by each invertebrate group in the model was derived from invertebrate diet information reported by Merritt and Cummins (1996). We determined the diet for each family, and the overall diet for each taxonomic group was then estimated proportionally based on the relative abundance of each family within the taxonomic group. We used EwE default values and the values from Meyer and Poepperl (2004) for invertebrate P/Q .

Fish.—Because stream-type (spring) Chinook Salmon in the Snake River basin are endangered, the number of individuals that can be captured under state and federal permits is limited. It is illegal to continue electroshocking surveys for any fish in a Chinook Salmon rearing stream after the stated limit of salmon has been reached; therefore, classic multiple-pass population estimates could not be conducted. Similarly, due to handling restrictions, we could not mark or fin-clip individuals for mark–recapture population estimates. Fish surveys were conducted by snorkeling in each stream and counting all of the fish. All snorkel surveys were performed by individuals who were trained and experienced in surveying and counting fish with this technique. A 200-m reach of stream was surveyed, and all individuals were counted. Fish length was estimated based on the size of the individual relative to rulers on divers' gloves.

The snorkel counts represented a minimum number of fish in a given reach. The actual number of individuals may be much greater than the number counted, especially for cryptic, benthic, and nocturnal fishes such as sculpins *Cottus* spp. Assessments of snorkel survey success

for salmonids suggest that for active pelagic fish, this method can be highly successful under clear water conditions in small- to mid-sized streams like the ones evaluated here, with snorkel survey estimates “capturing” 90% or more of the population (based on subsequent multiple-pass depletion estimates; Hankin and Reeves 1988; Orell and Erkinaro 2007). Others, however, have found that snorkel surveys for salmonids can substantially underestimate the abundance of fish in small streams (Thurow et al. 2006).

Salmonids were categorized into two size-classes for this study: smaller than 150 mm and larger than 150 mm. Given the near-ideal conditions for snorkel surveys in these systems (high water clarity, moderate gradient, limited large wood or wood jams, limited boulders, and moderate stream size), the detection of large salmonids in the stream snorkel surveys was assumed to be relatively high. We assumed 95% efficiency in our estimates; therefore, the point estimate used in the initial model for the biomass of adult salmonids and other nonbenthic fish larger than 150 mm (e.g., whitefishes) was therefore increased by only 5% above the estimate based on observations alone. The estimated biomass based on observation alone for these large salmonids was derived from the total number and estimated size of observed individuals during the snorkel surveys. Because each individual observed in the snorkel survey had an associated length estimate, we could apply mass–length relationships developed from the captured individuals in these and similar streams nearby with comparable conditions to determine an approximate mass for each larger fish observed. The total estimated biomass based on observations alone was then calculated as the sum of all individual mass estimates. This value was increased by 5% to account for the assumed 95% (rather than 100%) capture efficiency. The upper bound of the biomass estimate for larger salmonids assumed 90% efficiency and increased the biomass observed by 10%. The lower bound for the biomass estimate assumed 100% efficiency and therefore used the summed total alone. The biomass point estimate for the initial model (i.e., before balancing) was therefore 10% greater than the estimated biomass based on observed fish. The lower bound for the estimated biomass of smaller salmonids assumed 100% capture efficiency and was therefore based on the observed number of individuals multiplied by a biomass estimate for each individual based on the fish’s estimated length and an established mass–length relationship for that species in the system. The upper bound of the biomass estimate for smaller salmonids assumed only 80% capture efficiency and added 20% to the observed biomass estimates.

Snorkel surveys for benthic species, such as sculpins and darters, are much less efficient than snorkel surveys for salmonids and other pelagic species (Ensign et al. 1995; Williams et al. 2004). Sculpins were prevalent in the study streams and were a common diet item for large Brook Trout. We assumed only 30% efficiency in capturing sculpins during the snorkel surveys. When observed, sculpin lengths were estimated during the snorkel surveys, and a subset of individuals was captured during the subsequent electrofishing efforts in which salmonids were collected for mass-length relationships and diet. Therefore, as with salmonids, the estimated biomass of each individual sculpin was estimated from mass-length relationships, and the initial total biomass observations were calculated by summing the estimated biomass of all individuals. The upper bound of sculpin biomass assumed only 10% efficiency in the snorkel surveys, and the lower bound assumed 50% efficiency in the snorkel surveys.

Production estimates for salmonids in these systems were based on abundance estimates, mean fish biomass estimates, and documented growth rates of Brook Trout, Rainbow Trout, and Chinook Salmon from nearby streams, where fish were captured and individually marked during early summer and then recaptured during mid-summer and late summer. Growth (g/d) was estimated based on the natural logarithm of the change in mass over the time between capture events. These growth rate estimates were multiplied by mean biomass (DW) of fish in each species and age-group to get a mean production rate per fish. The mean production rate was then multiplied by the estimated abundance of each species and age-group. Bull Trout growth rates were assumed to be similar to those of Brook Trout because the Brook Trout is the only other char species in the study stream. Growth rates of the Cutthroat Trout were assumed to be similar to those of its congener, the Rainbow Trout (both in genus *Oncorhynchus*). No Chinook Salmon larger than 150 mm were marked and subsequently recaptured in any of the nearby study streams. For both Brook Trout and Rainbow Trout, the growth rate of fish larger than 150 mm was a bit less than one-half the growth rate for smaller fish. We therefore estimated the growth rate of larger individuals (>150 mm) to be 40% of the growth rate for smaller individuals. With no growth rate data and no comparable species for sculpins or whitefishes, we used the mean growth rate of salmonid species from the appropriate size-classes to estimate production of the sculpin and whitefish groups. We recognize that the actual growth rates may differ, but we deemed these to be reasonable estimates at this stage. Because overall production was calculated from growth rates, mean biomass, and fish abundance for each group, all of which were

estimated values, the error around production estimates was large. To constrain the range in balancing the model, we assumed that growth rates and mean fish biomass were accurate and we used the variability in fish abundance estimates to estimate the range in productivity ($\text{g}\cdot\text{m}^{-2}\cdot\text{d}^{-1}$) for fish in each reach. Given the uncertainty in both abundance and growth rate estimates for sculpins, we applied sculpin productivity estimates that exceeded 10% of the point estimate when balancing the model in Ecopath.

Fish diets for each species and size-class were based on (1) direct data from fish of that species and size-class collected in SFS or other streams in the Salmon River basin, (2) the diet of a given species taken as reported in the literature from systems outside the Salmon River basin, and (3) the diet of related species within SFS or other Salmon River basin streams. Terrestrial subsidies are not well addressed in the EwE model given its need for a balance of input and output. We accounted for terrestrial inputs through an “external sources” diet item, which allowed fish to obtain food from an external source that did not require a source from within the system. We also used the external source pool to account for migration of juvenile Chinook Salmon. Because most juvenile Chinook Salmon are only in the system for the summer, we prorated their diet to one-third of its actual value for each diet item and made up the difference in the external source category. This accounts for the impacts of juvenile Chinook Salmon for 4 months of the year and allows for their migration from the system.

We had good empirical diet data for juvenile Chinook Salmon, Brook Trout, and Rainbow Trout during summer 2004. We had limited but usable diet data for sculpins in this system, but data from other systems were considered in developing the sculpins’ diet (Zimmerman and Vondracek 2007). Diet data for Cutthroat Trout, Bull Trout, and whitefishes were not available from 2004 surveys. Cutthroat Trout were assumed to have a diet similar to that of Rainbow Trout (i.e., the other resident *Oncorhynchus* species), and the Bull Trout diet was assumed to be similar to the diet of Brook Trout (the other *Salvelinus* species). The whitefishes’ diet was assumed to consist primarily of invertebrates (limited piscivory), and consumption by the larger fish captured was assumed to reflect a broad spectrum of invertebrate taxonomic groups (Whiteley 2007). Whitefishes represented a small proportion of the overall fish biomass; thus, error in their diet had limited influence on the overall model outputs. We used EwE default values and values from Harvey and Kareiva (2005) for fish P/Q .

BALANCING THE ECOPATH MODEL

Once all of the data were entered into the Ecopath data files, the model was tested for “balance”—that is, whether production met demand for each taxonomic and resource group. The original data were out of balance, and we used the recommendations of Coll et al. (2009) to manually adjust the model to bring it into balance (rather than using any automated balancing tools). Our best data (periphyton biomass, invertebrate biomass, fish biomass for salmonids, and fish diet) were assumed to be the most accurate; therefore, when a given taxonomic group was out of balance, the metrics with a greater degree of uncertainty were adjusted first. For all invertebrate taxonomic groups, point estimates were based on the mean of three Hess samples. In balancing the model, changes in biomass and productivity for each taxonomic group were restricted to values within the range from the three samples. If adjustments in diet were implemented, they constituted no more than a 10% change in any one diet item. Sculpins were particularly “out of balance” in this system and could only be balanced with a standing biomass far greater than what was observed. Given the particularly low detection of sculpins in the snorkel surveys during which fish were counted, this is an understandable adjustment. We were never able to bring the detrital pool in this model into balance. The detrital pool can be difficult to balance in Ecopath models, especially for systems that receive considerable inputs of detritus from external sources, as is common for forested streams. Lack of balance in the detrital pool does not influence the other groups, and given the nature of streams and the high abundance of detrital input, we performed the Ecosim model analysis with detritus out of balance. The final Ecopath model is presented in Table 1, and diet data are presented in Table 2.

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