

Bioaccumulation and Aquatic System Simulator (BASS) User's Manual Version 2.2

by

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Notice

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Foreword

This report describes the theoretical development, parameterization, and application software of the BASS **B**ioaccumulation and **A**quatic **S**ystem **S**imulator. This generalized, community-based simulation model is designed to predict the population and bioaccumulation dynamics of age-structured fish communities exposed to hydrophobic organic chemicals and class B and borderline metals that complex with sulfhydryl groups (e.g., cadmium, copper, lead, mercury, nickel, silver, and zinc). This report is not a case study on the application of BASS but a reference and user's guide. The intended audience of this report and associated software includes research fisheries ecologists, bioaccumulation researchers, and EPA environmental scientists and ecologists who must routinely analyze and estimate bioaccumulation of chemicals in fish for ecological or human health exposure assessments.

Although BASS has not been extensively field-tested, its process-based algorithms for predicting chemical bioaccumulation, growth of individual fish, predator-prey interactions, and population dynamics either have been corroborated or have been formulated using widely accepted ecological and ecotoxicological principles. Even when a process-based model has undergone only limited field testing, it can be an extremely useful tool. Process-based models enable users to observe quantitatively the results of a particular abstraction of the real world. Moreover, such models can be argued to be the only objective method to make extrapolations to unobserved or unobservable conditions. If the conceptualization and construction of process-based models are both comprehensive (i.e., holistic) and reasonable, then their output, validated or not, can still be used for comparative analyses. A model's ability to simulate trends and comparative dynamics are, in fact, often more important measures of a model's utility than is its ability to replicate a specific field or laboratory study. Although BASS can be used to analyze results from actual field studies, its principal intended use is to predict and compare the outcomes of alternative management options associated with pollution control, ecosystem management, or restoration activities.

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Abstract

BASS (**B**ioaccumulation and **A**quatic **S**ystem **S**imulator) is a Fortran 95 simulation program that predicts the population and bioaccumulation dynamics of age-structured fish assemblages exposed to hydrophobic organic pollutants and class B or borderline metals that complex with sulfhydryl groups (e.g., cadmium, copper, lead, mercury, nickel, silver, and zinc). The model's bioaccumulation algorithms are based on diffusion kinetics and are coupled to a process-based model for the growth of individual fish. These algorithms consider both biological attributes of fishes and physico-chemical properties of the chemicals that together determine diffusive exchange across gill membranes and intestinal mucosa. Biological characteristics used by the model include the fish's gill morphometry, feeding and growth rate, and proximate composition (i.e., its fractional aqueous, lipid, and structural organic content). Relevant physico-chemical properties includes the chemical's aqueous diffusivity, n-octanol / water partition coefficient (K_{ow}), and, for metals, binding coefficients to proteins and other organic matter. BASS simulates the growth of individual fish using a standard mass balance, bioenergetic model (i.e., growth = ingestion - egestion - respiration - specific dynamic action - excretion). A fish's realized ingestion is calculated from its maximum consumption rate adjusted for the availability of prey of the appropriate size and taxonomy. The community's food web is delineated by defining one or more foraging classes for each fish species based on its body weight, body length, or age. The dietary composition of each of these foraging classes is specified as a combination of benthos, incidental terrestrial insects, periphyton / attached algae, phytoplankton, zooplankton, and one or more fish species. Population dynamics are generated by predatory mortalities defined by the community's food web and standing stocks, physiological mortality rates, the maximum longevity of species, toxicological responses to chemical exposures, and dispersal. The model's temporal and spatial scales are that of a day and of a hectare, respectively.

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1. Introduction

Fish health can be defined from both an ecological and a human health / value perspective in many ways. Questions relating to fish health from an ecological perspective often include:

- 1) Are individual fish growth and condition sufficient to enable them to survive periods of natural (e.g., overwintering) and man induced stress?
- 2) Are individual fish species able to maintain sustainable populations? For example, is individual growth adequate for fish to attain the minimum body size required for reproduction? Is there adequate physical environment for successful spawning? Is there adequate physical habitat for the survival of the young-of-year?
- 3) Do regional fish assemblages exhibit their expected biodiversity or community structure based on biogeographical and physical habitat considerations?
- 4) Are regional fish assemblages maintaining their expected level of productivity based on biogeographical and physical habitat considerations?
- 5) Are appropriately sized fish abundant enough to maintain piscivorous wildlife (e.g., birds, mammals, and reptiles) during breeding and non-breeding conditions?
- 6) Are potential fish prey sufficiently free of contaminants (endocrine disruptors, heavy metals, etc.) so as not to interfere with the growth and reproduction of piscivorous wildlife?
- 7) How will native fishes respond to the introduction of nonnative fish species, including those stocked for recreational fishing?

From a human health or use perspective, another important question related to fish health is:

- 8) Is the fish community / assemblage of concern fishable? That is, are target fish species sufficiently abundant and of the desired quality? Fish quality in this context is often defined in terms of desired body sizes (e.g., legal or trophy length) and the absence of chemical contaminants.

Some of the important metrics or indicators that have often been used to assess such questions include: 1) physical habitat dimensions, e.g., bottom type and cover, occurrence of structural elements such as woody debris or sand bars, mean and peak current velocities, water temperature, sediment loading, etc., 2) community species and functional diversity, 3) total community biomass (kg/ha or kg/km), 4) the population density (fish/ha or fish/km) or biomass (kg/ha or kg/km) of the community's dominant species, 5) the age or size class structure of the community's dominant species, 6) annual productivity of the community and its dominant species, 7) individual growth rates

or condition factors (i.e., the fish's current body weight normalized to an expected body weight based on its current length), and 8) levels of chemical contaminants in muscle or whole fish.

From the perspective of evaluating alternative management options or of assessing expected future consequences of existing conditions, simulation models that can predict the individual and population growth of fish and their patterns of chemical bioaccumulation are important tools for analyzing several of the aforementioned dimensions of fish health.

Although the growth of individual fish has often been described using empirical models such as the von Bertalanffy, logistic, Gompertz, or Richards models (see for example Ricker (1979) and Schnute (1981)), process-based bioenergetic models such as those described by Kitchell et al. (1977), Minton and McLean (1982), Stewart et al. (1983), Cuenca et al. (1985), Stewart and Binkowski (1986), Beauchamp et al. (1989), Stewart and Ibarra (1991), Lantry and Stewart (1993), Rand et al. (1993), Roell and Orth (1993), Hartman and Brandt (1995a), Petersen and Ward (1999), Rose et al. (1999), Schaeffer et al. (1999), and van Nes (2002) are becoming the models of choice for predicting the growth of fish. Because these process-based models predict fish growth based on the mass or energy balance of ingestion, egestion, respiration, specific dynamic action, and excretion, they can generally be parameterized independently of their current application. Moreover, because of the inherent difficulties in obtaining reliable field-based measurements of fish population dynamics and productivity, researchers are increasingly using such bioenergetic models to characterize these population and community level endpoints. See for example Stewart and Ibarra (1991) and Roell and Orth (1993).

The ability to predict accurately the bioaccumulation of chemicals in fish has become an essential component of ecological and human health risk assessments for chemical pollutants. Not only are accurate estimates needed to predict realistic dietary exposures to humans and piscivorous wildlife, but such estimates are also needed to assess more accurately potential ecological risks to fish assemblages themselves. Although exposure-referenced benchmarks such as LC₅₀ and EC₅₀ have been widely used for hazard assessments, most deleterious effects of chemical pollutants are caused by the internal accumulation of those compounds, rather than their environmental concentrations per se. Many authors (Neely 1984, Friant and Henry 1985, McCarty et al. 1985, McCarty 1986, Connell and Markwell 1992, McCarty and Mackay 1993, Verhaar et al. 1995, van Loon et al. 1997) have discussed the benefits of explicitly considering chemical bioaccumulation

when assessing expected ecological consequences of chemical pollutants in aquatic and marine ecosystems. Residue-based toxicity studies confirm this supposition (Oppenhuizen and Schrap 1988, van Hoogen and Oppenhuizen 1988, Donkin et al. 1989, Tas et al. 1991, van Wezel et al. 1995, Driscoll and Landrum 1997).

Although concentrations of moderately hydrophobic chemicals in fish can often be predicted accurately assuming equilibrium partitioning of the chemicals between the fish's organic constituents and the aqueous environment, this approach frequently fails to predict observed concentrations of extremely hydrophobic chemicals and metals that are often the chemicals of greatest concern. Observed deviations can be either considerably above or below those predicted by equilibrium partitioning. Several factors can be identified to explain these discrepancies.

Lower than expected contamination levels can result when the length of exposure is insufficient to allow chemicals to equilibrate. Because bioconcentration and bioaccumulation are generally treated as linear, first order processes, the time needed for chemicals to equilibrate between fish and their exposure media is an increasing function of the elimination half lives of those chemicals in fish. For example, the time required for chemicals to achieve 95% of their equilibrium concentrations is approximately 4.3 times their elimination half lives. Because the elimination half lives of chemicals generally increase as their hydrophobicities increase, the time needed for chemicals to reach their equilibrium concentrations in fish also increases as a function of chemical hydrophobicity. Consequently, for extremely hydrophobic chemicals such as polychlorinated biphenyls (PCBs) and dioxins that have elimination half lives ranging from months to over a year, the time to equilibrium can be on the order of years. If the fish species of concern is short lived, the time needed for equilibrium can exceed the species expected life span. Even when time is sufficient for equilibration, whole-body concentrations of fish can be much lower than that expected from thermodynamic partitioning due to physical dilution of the chemical that accompanies body growth or due to in situ biotransformation of the parent compound.

One of two possible assumptions is implicitly made whenever equilibrium-based estimators are used. The first of these assumptions is that only the selected reference route of exposure is significant in determining the total chemical accumulation in fish. The alternative assumption is that there are multiple routes of exposure that all covary with the chosen reference pathway in a constant manner. For bioconcentration factors (BCFs), the implicit assumption is that virtually the entire burden is exchanged directly with the water across the fish's gills or possibly across its skin. Although direct aqueous uptake is

certainly the most significant route of exchange for moderately hydrophobic chemicals, dietary uptake accounts for most of a fish's body burden for extremely hydrophobic chemicals. This shift in the relative significance of the direct aqueous versus the dietary pathway is determined by the relative rates of exposure via these media and by a fundamental difference in the nature of chemical exchange from food and water. Consider, for example, the relative absolute exposures to a fish via food and water. The fish's direct aqueous exposure, AE ($\mu\text{g/d}$), is the product of its ventilation volume, Q (ml/d), and the chemical's aqueous concentration, C_w ($\mu\text{g/ml}$). Similarly, the fish's dietary exposure, DE ($\mu\text{g/d}$), is the product of its feeding rate, F_w (g wet wt/d), and the chemical's concentration in the fish's prey, C_p ($\mu\text{g/g wet wt}$). If the fish feeds only on one type of prey that has equilibrated with the water, one can calculate when the fish's aqueous and dietary exposures are equal using the equations

$$\begin{aligned} AE &= DE \\ Q C_w &= F_w C_p \\ Q/F_w &= BCF \end{aligned} \tag{1.1}$$

Using data from Stewart et al. (1983) and Erickson and McKim (1990), the ventilation-to-feeding ratio for a 1 kg trout would be on the order of $10^{4.3}$ ml/g. Assuming the quantitative structure activity relationship (QSAR) for the trout's prey is $BCF = 0.048 K_{ow}$ (Mackay 1982), one would conclude that food is the trout's predominant route of exposure for any chemical whose octanol / water partition coefficient is greater than $10^{5.6}$. For extremely hydrophobic chemicals, not only will fish be more exposed via food but they probably will assimilate chemicals from food more effectively than from the water. Although chemical exchange from both food and water occur by passive diffusion, uptake from food, unlike direct uptake from water, does not necessarily relax the diffusion gradient into the fish. This fundamental difference results from the digestion and assimilation of food that can actually cause the chemical concentrations of the fish's gut contents to increase (Connolly and Pedersen 1988, Gobas et al. 1988). Predicting residue levels for chemicals whose principal route of exchange is dietary is further complicated since most fish species demonstrate well-defined size-dependent, taxonomic, and temporal trends regarding the prey they consume. Consequently, one would not generally expect a single BAF to be sufficiently accurate for risk assessments for all fish species or even different sizes of the same species.

Process-based models that describe the kinetic exchange of chemicals from food and water in concert with the growth of fish provide objective and scientifically sound frameworks that can overcome many of the aforementioned limitations of equilibrium-based BAFs and BCFs. Although numerous models have been developed toward this end (Norstrom et al. 1976, Thomann 1981, Jensen et al. 1982, Thomann and Connolly 1984, Barber

et al. 1987, Gobas et al. 1988, Barber et al. 1991, Thomann et al. 1992, Gobas 1993, Madenjian et al. 1993), these models differ significantly regarding how food web structure and dietary exposures are represented.

This report describes the theoretical framework, parameterization, and use of BASS (**B**ioaccumulation and **A**aquatic **S**ystem **S**imulator). This generalized, process-based, Fortran 95 simulation model is designed to predict the growth of individuals and populations within an age-structured fish community and the bioaccumulation dynamics of those fish when exposed to

mixtures of metals and organic chemicals. The model is formulated such that its parameterization does not rely upon calibration data sets from specific toxicokinetic and population field studies, but rather upon physical and chemical properties that can be estimated using chemical property calculators such as CLOGP (<http://www.biobyte.com/bb/prod/clogp40.html>), or SPARC (Carreira et al. 1994) (also see <http://ibmlc2.chem.uga.edu/sparc/style/welcome.cfm>) and on ecological, morphological, and physiological parameters that can be obtained from the published literature or computerized databases.

2. Model Formulation

To model the chemical bioaccumulation and the growth of individuals and populations within an age-structured fish community, BASS solves the following system of differential equations for each age class or cohort of fish:

$$\frac{dB_f}{dt} = J_g + J_i - J_{bt} \quad (2.1)$$

$$\frac{dW_d}{dt} = F_d - E_d - R - EX - SDA \quad (2.2)$$

$$\frac{dN}{dt} = -EM - NM - PM \quad (2.3)$$

where B_f and W_d are the chemical body burden ($\mu\text{g}/\text{fish}$) and dry body weight (g dry wt/fish), respectively, of the average individual within the cohort; and N is the cohort's population density (fish/ha). In Equation (2.1), J_g is the net chemical exchange ($\mu\text{g}/\text{d}$) across the fish's gills from the water; J_i is the net chemical exchange ($\mu\text{g}/\text{d}$) across the fish's intestine from food; and J_{bt} is the chemical's biotransformation rate ($\mu\text{g}/\text{d}$). In Equation (2.2), F_d , E_d , R , EX , and SDA are the fish's feeding, egestion, routine respiration, excretion, and specific dynamic action (i.e., the respiratory expenditure in excess of R required to assimilate food), respectively, in units of g dry wt/d. Although many physiologically based models for fish growth are formulated in terms of energy content and flow (e.g., kcal/fish and kcal/d), Equation (2.2) is fundamentally identical to these bioenergetic models since the energy densities of fish depend on their dry weight (Kushlan et al. 1986, Hartman and Brandt 1995b, Schreckenbach et al. 2001). Finally, in Equation (2.3) EM , NM and PM are the cohort's rates (fish/ha/d) of emigration/dispersal, non-predatory, and predatory mortality, respectively. Although immigration can be a significant process in determining population sizes, this process is not presently modeled in BASS. Because cohort recruitment is treated as a boundary condition, the right-hand side of Equation (2.3) does not require a term for recruitment. Though it may not be immediately apparent from the notation used, these equations are tightly coupled to one another. For example, the realized feeding of fish depends on the availability (i.e., density and biomass) of suitable prey. The fish's predatory mortality, in turn, is determined by the individual feeding levels and population densities of its predators. Finally, the fish's dietary exposure is determined by its rate of feeding and the levels of chemical contamination in its prey.

The following sections describe how each mass flux in the above system of equations is formulated in BASS. **Table 2.1** summarizes the definitions of the variables used to develop these equations.

Because the system of units used to formulate chemical exchanges is essentially the CGS-system (centimeter, gram, second) and the system of units used to formulate a fish's growth is the CGD-system (centimeter, gram, day), some units conversion is necessary to make the coupled system of equations dimensionally consistent. The reader should also note that whereas the growth of fish is modeled in terms of dry weight, a fish's chemical bioaccumulation is formulated in terms of its live body weight since BASS models chemical uptake and excretion by fish as chemical diffusion between aqueous phases.

2.1. Modeling Internal Distribution of Chemicals

Chemical exchanges across gills of fish and from their food are generally considered to occur by passive diffusion of chemicals between a fish's internal aqueous phase and its external aqueous environment, whether the latter is the surrounding ambient water or the aqueous phase of the fish's intestinal contents. Consequently, to model these exchanges one must first consider how chemicals distribute within the bodies of fish. If a fish is conceptualized as a three-phase solvent consisting of water, lipid, and non-lipid organic matter, then its whole-body chemical concentration can be expressed as

$$C_f = \frac{B_f}{W_w} = P_a C_a + P_l C_l + P_o C_o \quad (2.4)$$

$$= \left(P_a + P_l \frac{C_l}{C_a} + P_o \frac{C_o}{C_a} \right) C_a$$

where W_w is the fish's live weight (g wet wt/fish); P_a , P_l , and P_o are the fractions of the whole fish that are water, lipid, and non-lipid organic material, respectively; and C_a , C_l , and C_o are the chemical's concentrations in those respective phases. Because the depuration rates of chemicals from different fish tissues often do not differ significantly (Grzenda et al. 1970, van Veld et al. 1984, Branson et al. 1985, Norheim and Roald 1985, Kleeman et al. 1986a, b), internal equilibration between these three phases can be assumed to be rapid in comparison to external exchanges. For organic chemicals, this assumption means that Equation (2.4) simplifies to

$$C_f = (P_a + P_l K_l + P_o K_o) C_a \quad (2.5)$$

where K_l and K_o are the chemical's partition coefficients between lipid and water and between organic carbon and water, respectively.

For metals, however, Equation (2.4) is more complicated. Although metals do partition into lipids (Simkiss 1983), their accumulation within most other organic media occurs by

complexation reactions with specific binding sites. Consequently, for metals the term $P_o C_o / C_a$ in Equation (2.4) could be formulated as a function of an appropriate stability coefficient and the availability of binding sites. Appendix A summarizes an equilibrium complexation model that was initially formulated for BASS. Despite its apparent correctness, this algorithm greatly overestimated metal (in particular mercury) bioaccumulation in fish. Although this overestimation can be attributed to several factors, the most likely explanation for the algorithm's unsatisfactory performance is that kinetics limits the complexation of metal in fish. Because kinetic modeling was considered incongruent with the time scales of most of the other major processes represented elsewhere in BASS, a much simpler algorithm was adopted.

Because many fate and transport models (e.g., EXAMS and WASP) have successfully used operationally defined distribution coefficients K_d to model the accumulation of metals in organic media, a similar approach was adopted for BASS. Thus, for a metal

$$C_f = (P_a + P_l K_l + P_o K_d) C_a \quad (2.6)$$

where K_l is again an appropriate partition coefficient between lipid and water; and K_d is an appropriate metal-specific distribution coefficient. Although this equation appears identical to Equation (2.5) for organic contaminants, the relative values of K_d and K_o in relation to K_l can be remarkably different. See Section 3.1.

Because C_a equals the chemical's ambient environmental water concentration C_w at equilibrium, it follows from Equations (2.4) and (2.6) that a fish's thermodynamic bioconcentration factor ($K_f = C_f / C_w$ at equilibrium) for a chemical pollutant of concern is

$$K_f = \begin{cases} P_a + P_l K_l + P_o K_o & \text{for organics} \\ P_a + P_l K_l + P_o K_d & \text{for metalics} \end{cases} \quad (2.7)$$

2.2. Modeling Exchange from Water

Because chemical exchange across the gills of fish occurs by simple diffusion, such exchanges can be modeled by Fick's first law of diffusion as

$$J_g = S_g k_g (C_w - C_a) \quad (2.8)$$

where S_g is the fish's total gill area (cm^2); k_g is the chemical's conductance (cm/s) across the gills from the interlamellar water; and C_w is the chemical's concentration ($\mu\text{g/ml}$) in the environmental water (Yalkowsky et al. 1973, Mackay 1982, Mackay and Hughes 1984, Gobas et al. 1986, Gobas and Mackay 1987, Barber et al. 1988, Erickson and McKim 1990). When Equations (2.5), (2.6), and (2.7) are substituted into this equation,

one then obtains

$$J_g = S_g k_g \left(C_w - \frac{C_f}{K_f} \right) \quad (2.9)$$

Although the chemical's conductance k_g could be specified as a ratio of the chemical's diffusivity to the thickness of an associated boundary layer, implementation of this definition can be problematic since the boundary layer thickness is a function of the gill's ventilation velocity and varies along the length of the gill's secondary lamellae. To avoid this problem, a fish's net chemical exchange rate coefficient, $S_g k_g$, can be estimated by reformulating the gill's net chemical exchange as

$$J_g = Q (C_w - C_B) \quad (2.10)$$

where Q is the fish's ventilation volume (cm^3/s); and C_B is the chemical's bulk concentration in the expired gill water. When Equations (2.8) and (2.10) are equated, it follows that

$$S_g k_g = Q \left(\frac{C_w - C_B}{C_w - C_a} \right) \quad (2.11)$$

Despite its appearance, the right-hand side of this equation can be readily quantified. In particular, the ventilation volume of fish can be estimated by

$$Q = \frac{R_{O_2}}{\alpha_{O_2} C_{w,O_2}} \quad (2.12)$$

where R_{O_2} is the fish's rate of oxygen consumption ($\mu\text{g/s}$); α_{O_2} is the fish's oxygen assimilation efficiency; and C_{w,O_2} is the environmental water's dissolved oxygen concentration ($\mu\text{g/ml}$). If one makes certain assumptions concerning the geometry of the interlamellar spaces and the nature of mass transport between the gill's secondary lamellae, the chemical's normalized bulk concentration in the expired gill water ($(C_w - C_B) / (C_w - C_a)$) can also be calculated as outlined in the following.

Because the gill's secondary lamellae form flat channels having high aspect ratios (i.e., mean lamellar height / interlamellar distance), they can be treated as parallel plates, and the flow of water between them can be treated as Poiseuille slit flow (Hills and Hughes 1970, Stevens and Lightfoot 1986). Under this assumption, an expression for a chemical's concentration in the bulk expired gill water can be obtained using the solutions of the partial differential equation (PDE) that describes steady-state, convective mass transport between parallel plates, i.e.,

$$\frac{3}{2} \left(1 - \frac{x^2}{r^2} \right) V \frac{\partial C}{\partial y} = D \frac{\partial^2 C}{\partial x^2} \quad (2.13)$$

where V is the gill's mean interlamellar flow velocity (cm/s); D is the chemical's aqueous diffusivity (cm^2/s); and x and y are the

lateral and longitudinal coordinates of the channel along which diffusion and convection occurs, respectively. In this equation, $C = C(x, y)$ is the chemical's interlamellar concentration at the distances x from the surface of the lamellae and y along its length. The surfaces of adjacent lamellae are located at $x = \pm r$ where r is the hydraulic radius of the lamellar channel that equals half the interlamellar distance d (cm). The midline between adjacent lamellae is therefore denoted by $x=0$. The gill's mean interlamellar flow velocity can be readily formulated as the ratio of the fish's ventilation volume to the gill's cross sectional pore area, X_g (cm²). Because the gill's pore area is related to its lamellar surface area by

$$X_g = \frac{S_g d}{l} \quad (2.14)$$

where d is the mean interlamellar distance (cm); and l is the mean lamellar length (cm) (Hills and Hughes 1970), a fish's mean interlamellar flow velocity is given by

$$V = \frac{Ql}{S_g d} \quad (2.15)$$

To solve Equation (2.13), two boundary conditions must be specified. Because adjacent lamellae presumably exchange the chemical equally well, the solutions should be symmetrical about the channel's midline. To insure this characteristic, the boundary condition

$$\left. \frac{\partial C}{\partial x} \right|_{x=0} = 0 \quad (2.16)$$

is assumed. The second necessary boundary condition must describe how chemical exchange across the secondary lamellae actually occurs. Assuming steady state diffusion from the interlamellar water to the fish's aqueous blood, this boundary condition can be formulated as

$$D \left. \frac{\partial C}{\partial x} \right|_{x=r} = -k_m [C(r, y) - C_a] \quad (2.17)$$

where k_m is the permeability (cm/s) of the gill membrane. Although this boundary condition has been used as is (Barber et al. 1991), it can also be modified to address potential perfusion limitations on gill uptake. To accomplish this latter task, a formulation patterned after Erickson and McKim (1990) is used. In particular, consider the following reformulation

$$D \left. \frac{\partial C}{\partial x} \right|_{x=r} = -k_m \left\{ C(r, y) - \left[C_a(l) + \frac{J_s(y, l)}{q_p} \right] \right\} \quad (2.18)$$

where $C_a(y)$ is the chemical's aqueous phase concentration at point y along the length of a secondary lamella; $C_a(l) = C_a$ is the

chemical's concentration in the afferent lamellar blood; $J_s(y, l)$ is the chemical's accumulated uptake ($\mu\text{g/s}$) along the lamellar segment $[y, l]$; and q_p is the lamellar perfusion rate (cm³/s). If both sides of the lamella uptake chemical, then $J_s(y, l)$ can be formulated as

$$\begin{aligned} J_s(y, l) &= 2 \int_y^l \int_0^h D \left. \frac{\partial C}{\partial x} \right|_{x=r} du dv \\ &= 2 h D \int_y^l \left. \frac{\partial C}{\partial x} \right|_{x=r} dv \end{aligned} \quad (2.19)$$

where h is the height (cm) of the secondary lamella. Using this expression, the boundary condition (2.18) can now be written as

$$D \left. \frac{\partial C}{\partial x} \right|_{x=r} = -k_m \left[C(r, y) - C_a - \frac{2 h D}{q_p} \int_y^l \left. \frac{\partial C}{\partial x} \right|_{x=r} dv \right] \quad (2.20)$$

Once the solution of Equation (2.13) for these boundary conditions has been obtained, the chemical's bulk concentration in the expired gill water can be evaluated using the weighted average

$$C_B = \frac{\int_0^r C(x, l) \left(1 - \frac{x^2}{r^2} \right) dx}{\int_0^r \left(1 - \frac{x^2}{r^2} \right) dx} \quad (2.21)$$

that scales each concentration profile $C(x, l)$ by its relative velocity.

A canonical solution to Equation (2.13) can be obtained by nondimensioning $C(x, y)$, x , and y as follows

$$\Theta = \frac{C - C_a}{C_w - C_a} \quad (2.22)$$

$$X = \frac{x}{r} \quad (2.23)$$

$$Y = \frac{yD}{Vr^2} \quad (2.24)$$

When this is done, the chemical's dimensionless bulk concentration is given by

$$\Theta_B = \frac{C_B - C_a}{C_w - C_a} = \frac{\int_0^1 \Theta(X, N_{Gz}) (1 - X^2) dX}{\int_0^1 1 - X^2 dX} \quad (2.25)$$

where $N_{Gz} = (l D)/(V r^2)$ is the gills' dimensionless lamellar length or Graetz number. Two important features of this expression can now be observed. First, one can easily verify that

$$1 - \Theta_B = \frac{C_w - C_B}{C_w - C_a} \quad (2.26)$$

Consequently, Equation (2.11) can be rewritten as

$$S_g k_g = Q(1 - \Theta_B) \quad (2.27)$$

Secondly, analytical expressions for Θ_B are readily available (Brown 1960, Grimsrud and Babb 1966, Colton et al. 1971, Walker and Davies 1974). In particular, a chemical's dimensionless bulk concentration can be evaluated by

$$\Theta_B = \sum_{m=0}^{\infty} B_m \exp(-\frac{2}{3} \lambda_m^2 N_{Gz}) \quad (2.28)$$

where the coefficients B_m and exponents λ_m are known functions of the gills' dimensionless conductance or Sherwood number

$$N_{Sh} = \frac{k_m r}{D} \quad (2.29)$$

and the fish's ventilation / perfusion volume ratio. See Appendix B. Although this infinite series solution does not have a convenient convergence formula, for Sherwood numbers and ventilation / perfusion ratios that are typical of fish gills, only the first two terms of the series are needed to estimate Θ_B with less than 1% error (see Barber et al. 1991). See **Figure 2.1** and **Figure 2.2** for displays of λ_1 and B_1 and of λ_2 and B_2 , respectively.

2.3. Modeling Exchange from Food

Chemical uptake from food has usually been modeled assuming that fish assimilate a constant fraction of the chemical that they ingest, i.e.,

$$J_i = \alpha_c C_p F_w \quad (2.30)$$

where α_c is the assimilation efficiency (dimensionless) for the chemical; C_p is the chemical's concentration ($\mu\text{g/g}$ wet wt) in the ingested prey; and F_w is the fish's daily wet weight prey consumption (g wet wt/d) (Norstrom et al. 1976, Jensen et al. 1982, Thomann and Connolly 1984, Niimi and Oliver 1987).

However, because the chemical exchange across the intestine is driven by diffusive gradients (Vetter et al. 1985, Clark et al. 1990, Gobas et al. 1993), such formulations are thermodynamically realistic only if α_c is a decreasing function of the fish's total body concentration C_f .

A thermodynamically based description for the dietary uptake of chemicals can be formulated using the simple mass balance relationship

$$J_i = C_p F_w - C_e E_w \quad (2.31)$$

where E_w is the fish's daily wet weight egestion (g wet wt/d) and C_e is the pollutant's chemical concentration ($\mu\text{g/g}$ wet wt) in the fish's feces. When this equation is reformulated in terms of dry weight feeding and egestion (i.e., $F_d = P_{dp} F_w$ and $E_d = P_{de} E_w$ where P_{dp} and P_{de} denote the prey's and feces' dry weight fractions, respectively) the fish's net dietary exchange becomes

$$\begin{aligned} J_i &= \frac{C_p F_d}{P_{dp}} - (C_{ae} E_a + C_{de} E_d) \\ &= \frac{C_p F_d}{P_{dp}} - \left(\frac{E_a}{E_d} + \frac{C_{de}}{C_{ae}} \right) C_{ae} E_d \\ &= \frac{C_p F_d}{P_{dp}} - \left(\frac{P_{ae}}{P_{de}} + \frac{C_{de}}{C_{ae}} \right) C_{ae} E_d \\ &= \frac{C_p F_d}{P_{dp}} - \left(P_{ae} + P_{de} \frac{C_{de}}{C_{ae}} \right) \frac{C_{ae} E_d}{P_{de}} \end{aligned} \quad (2.32)$$

where C_{ae} and C_{de} are the pollutant's chemical concentrations in the aqueous and dry phases of the fish's feces, respectively; E_a is the mass / volume of the feces' aqueous phase; and P_{ae} and P_{ap} are the aqueous fractions of the fish's feces and prey, respectively. To parameterize Equation (2.32), two assumptions are made.

Because the transit time through the gastrointestinal tract is relatively slow, the first of these assumptions is that the concentrations of chemicals in the fish's aqueous blood, intestinal fluids, and dry fecal matter equilibrate with one another. Consequently, $C_{ae} = C_a$. Moreover, for organic chemicals the concentration ratio C_{de} / C_{ae} can be replaced with an organic carbon / water partition coefficient, K_{oc} (e.g., Briggs 1981, Karickhoff 1981, Chiou et al. 1986), and for metals this ratio can be substituted with a distribution coefficient similar to the one used in Equation (2.6).

Although reported values for the percent moisture of the intestinal contents of fish vary typically between 50 and 80% (Brett 1971, Marais and Erasmus 1977, Grabner and Hofer

1985), the second assumption made to parameterize Equation (2.32) assumes that the fish's intestinal contents and whole body are osmotically equilibrated. Consequently, $P_{ae} = P_a$. If this assumption is reasonable, then meals with the same dry weights but different wet weights should be processed by the fish at equal rates and efficiencies since both will attain the same proximate composition relatively soon after ingestion. Having the same proximate composition implies that the concentrations of digestive enzymes acting on the meals will be comparable and that the physical forces exerted by the gut contents that control gastric mobility will also be comparable. Because Bromley (1980), Garber (1983), and Ruohonen et al. (1997) demonstrated that initial dietary moisture content had no significant effect on the assimilation efficiencies of turbot (*Scophthalmus maximus*) or on gastric evacuation rates of yellow perch (*Perca flavescens*) and rainbow trout (*Oncorhynchus mykiss*), respectively, the assumption that $P_{ae} = P_a$ seems reasonable.

Using the stated assumptions and Equations (2.5), (2.6), and (2.7), Equation (2.32) now can be rewritten as

$$J_i = \frac{C_p F_d}{P_{dp}} - [P_a + (1 - P_a)K_{oc}] \frac{C_f E_d}{K_f (1 - P_a)} \quad (2.33)$$

$$= \frac{C_p F_d}{P_{dp}} - \frac{K_e C_f E_d}{K_f (1 - P_a)}$$

where K_e is the distribution coefficient describing the chemical partitioning between the aqueous and dry organic matter phases of the fish's intestinal contents. Although BASS uses this equation to calculate a fish's net chemical dietary exchange, this equation can also be further manipulated as follows

$$J_i = \left[1 - \frac{P_{dp} C_f K_e E_d}{(1 - P_a) C_p K_f F_d} \right] \frac{C_p F_d}{P_{dp}} \quad (2.34)$$

$$= \left[1 - (1 - \alpha_f) \frac{P_{dp} C_f K_e}{P_a C_p K_f} \right] C_p F_f$$

where $\alpha_f = (F_d - E_d)/F_d$ is the fish's food assimilation efficiency (g dry wt assimilated/g dry wt ingested), and $P_a = (1 - P_a)$ is the fish's dry fraction. In other words, a thermodynamically based assimilation efficiency for Equation (2.30) corresponds to

$$\alpha_c = 1 - (1 - \alpha_f) \frac{P_{dp} C_f K_e}{P_a C_p K_f} \quad (2.35)$$

Thus, Equation (2.33) is equivalent to Equation (2.30) with a chemical assimilation efficiency that decreases as the fish's whole-body chemical burdens or concentrations increase. Studies by Lieb et al. (1974), Gruger et al. (1975), and Opperhuizen and Schrap (1988) corroborate this prediction (see Barber et al.

1991). Moreover, using in situ preparations of channel catfish intestines Doi et al. (2000) have clearly established that preexposures to 3,4,3',4'-tetrachlorobiphenyl do decrease intestinal uptake rates.

Although the preceding model development demonstrates the potential logical inconsistency between an assumed constant chemical assimilation efficiency model for dietary chemical uptake and a thermodynamically based model, many researchers continue to use the former assumption and model. Parameters for these constant chemical assimilation efficiency models generally have been estimated using the following equations proposed by Bruggeman et al. (1981)

$$\frac{dC_f}{dt} = \alpha_c \phi_{ww} C_p - k_2 C_f \quad (2.36)$$

$$C_f = \left(\frac{\alpha_c \phi_{ww} C_p}{k_2} \right) [1 - \exp(-k_2 t)] \quad (2.37)$$

where ϕ_{ww} is the fish's specific feeding rate (g wet wt/g wet wt/d); and k_2 is the fish apparent elimination rate coefficient (g wet wt/g wet wt/d or 1/d) that is the sum of its rate coefficients of growth (γ), biotransformation (k_{bt}), and actual excretion (k_{ex}). See for example Muir et al. (1992), Dabrowska et al. (1996), and Fisk et al. (1998). Unfortunately, many researchers have failed to acknowledge that Equation (2.37) is the solution to Equation (2.36) only when initial time is $t_0 = 0$ and the fish's initial whole-body concentration is zero (i.e., $C_f(t_0) = 0$). This fact, combined with the ability of Equation (2.37) to fit experimental results statistically, has been at least partially responsible for the perpetuation of the idea of constant chemical assimilation efficiencies.

The general solution to Equation (2.36) is actually

$$C_f = \frac{\alpha_c \phi_{ww} C_p}{k_2} + \left[C_f(t_0) - \frac{\alpha_c \phi_{ww} C_p}{k_2} \right] \exp[-k_2(t - t_0)] \quad (2.38)$$

When this solution is re-differentiated, one observes that

$$\frac{dC_f}{dt} = \left[\alpha_c \phi_{ww} C_p - k_2 C_f(t_0) \right] \exp[-k_2(t - t_0)] \quad (2.39)$$

Using this exponential form of Equation (2.36), one can analyze the parameter behavior of a dietary exposure during consecutive time segments for which ϕ_{ww} and C_p are constant. Therefore, let $T = (t_2 - t_0)$ denote the length of such a dietary exposure, and let t_1 denote the time when the exposure is half over. During the first half of the exposure (i.e., $t_0 < t < t_1$) the fish's bioaccumulation dynamics will be described by Equation (2.39). During the second half of the exposure, however, these dynamics will be

described by

$$\frac{dC_f}{dt} = [\hat{\alpha}_c \Phi_{ww} C_p - \hat{k}_2 C_f(t_1)] \exp[-\hat{k}_2(t - t_1)] \quad (2.40)$$

where $\hat{\alpha}_c$ and \hat{k}_2 are the fish's assimilation efficiency and apparent elimination rate coefficient that may require updating for $t_1 < t < t_2$. If an equation of the form of Equation (2.37) is assumed to describe the fish's bioaccumulation dynamics over the entire interval $[t_0, t_2]$, then the derivatives specified by Equations (2.39) and (2.40) must be equal when evaluated for $t = t_1$. This consistency condition, that is analogous to the preservation of derivatives that occurs when approximating a function with Bernstein polynomials, requires that

$$\hat{\alpha}_c \Phi_{ww} C_p - \hat{k}_2 C_f(t_1) = [\alpha_c \Phi_{ww} C_p - k_2 C_f(t_0)] \exp[-k_2(t_1 - t_0)] \quad (2.41)$$

which implies that

$$\hat{\alpha}_c = \frac{\hat{k}_2 C_f(t_1)}{\Phi_{ww} C_p} + \left[\alpha_c - \frac{k_2 C_f(t_0)}{\Phi_{ww} C_p} \right] \exp[-k_2(t_1 - t_0)] \quad (2.42)$$

When the fish's initial whole-body concentration is zero, this equation can be shown to reduce to

$$\hat{\alpha}_c = \alpha_c \left\{ \frac{\hat{k}_2}{k_2} + \left(1 - \frac{\hat{k}_2}{k_2} \right) \exp[-k_2(t_1 - t_0)] \right\} \quad (2.43)$$

This equation shows that unless $\hat{k}_2 = k_2$, chemical assimilation efficiencies estimated for different times and initial whole-body concentration will be different. Phrased another way, this equation implies that the fish's ability to excrete, biodilute, and biotransform chemicals, as measured by \hat{k}_2 and k_2 , contributes to the determination of the fish's realized chemical assimilation efficiencies. Because specific growth rates ($\gamma = W_w^{-1} dW_w/dt$) and chemical excretion rate coefficients (k_{ex}) for fish are generally related to the fish's body size as allometric power functions, i.e.,

$$\gamma = \alpha_1 W_w^{\alpha_2} \quad (2.44)$$

$$k_{ex} = \beta_1 W_w^{\beta_2} \quad (2.45)$$

where $\alpha_2 < 0$ and $\beta_2 < 0$ (Barber et al. 1988, Sijm et al. 1993, Sijm and van der Linde 1995, Sijm et al. 1995), one would expect that $\hat{k}_2 < k_2$ when significant growth occurs during the experiment. Consequently, one would also expect that $\hat{\alpha}_c < \alpha_c$. Importantly, this simple analysis is corroborated by findings of Ram and Gillet (1993) who showed that assimilation efficiencies for a variety of organochlorines by oligochaetes decreased as chemical exposures progressed.

In terms of application, the proposed fecal partitioning model (i.e., Equation (2.33)) is best suited to circumstances where its equilibrium assumptions are reasonably satisfied, as herein where the object is to predict the dietary exchange of the average individual of a population. A more kinetically based approach may be needed, however, when trying to describe the toxicokinetics of individual fish. See for example Nichols et al. (1998). Readers are referred to Barber (Barber 2008) for a more thorough discussion and analysis of dietary uptake algorithms used to model chemical bioaccumulation in fish.

2.4. Modeling Chemical Biotransformation

BASS assumes that the metabolism of xenobiotic chemicals in fish is a simple, first order reaction of the chemical's aqueous phase concentration, i.e.,

$$J_{bt} = (\epsilon_a C_a) (P_a W_w) \quad (2.46)$$

where ϵ_a is the fish's aqueous phase biotransformation rate coefficient (1/d); and $(P_a W_w)$ is the volume of the fish's aqueous phase. If Equations (2.9) and (2.46) are used to describe chemical bioconcentration during a water-only exposure without growth, then a fish's whole-body concentration would be modeled as

$$\begin{aligned} \frac{dC_f}{dt} &= \frac{1}{W_w} \frac{dB_f}{dt} = \frac{S_g k_g}{W_w} \left(C_w - \frac{C_f}{K_f} \right) - \frac{\beta P_a C_f}{K_f} \\ &= k_u C_w - (k_{ex} + k_{bt}) C_f \end{aligned} \quad (2.47)$$

where k_u , k_{ex} , and k_{bt} are the fish's rate coefficients of gill uptake, excretion, and biotransformation, respectively. In terms of quantitative structure activity relationships (QSARs), this equation predicts that a fish's whole-body biotransformation rate k_{bt} should be inversely proportional to its thermodynamic bioconcentration factor K_f that in turn is proportional to the chemical's K_{ow} . This relationship, however, will also be influenced by any QSAR dependencies that the fish's aqueous phase biotransformation rate ϵ_a might have. See de Wolf et al. (1992) and de Bruijn et al. (1993).

2.5. Modeling Temperature Effects on Physiological Rates

Because temperature affects a fish's feeding, assimilation, respiration, and egestion, a general discussion of how temperature modulates these processes is in order before describing how BASS actually models fish growth. Although the temperature dependence of physiological processes is often described using an exponential response equation, e.g.,

$$p_1 = p_0 \exp[\epsilon(T_1 - T_0)] \quad (2.48)$$

where p_0 and p_1 are the reaction rates of the process at temperatures T_0 and T_1 , respectively, such descriptions are generally valid only within a range of the organism's thermal tolerance. In most cases, the process's reaction rate increases exponentially with increasing temperature up to a temperature T_1 after which it decreases. Moreover, the temperature at which a process's rate is maximum is often very close to the organism's upper thermal tolerance limit. To model this behavior, Thornton and Lessem (1978) developed a logistic multiplier to describe the temperature dependence of a wide variety of physiological processes. Although this algorithm has been used successfully in many fish bioenergetic models, BASS uses an exponential-type formulation that responds hyperbolically to increasing temperature. Importantly, such algorithms can be easily parameterized.

Let P denote the rate of a physiological process, and let T_1 denote the temperature at which the rate is at its maximum value. If this process generally exhibits an exponential response to temperature changes well below T_1 , then Equation (2.48) can be used to describe this process for T and $T_0 \ll T_1$, i.e.,

$$P = P_0 \exp[\varepsilon(T - T_0)] \quad (2.49)$$

$$\frac{dP}{dT} = \varepsilon P \quad (2.50)$$

where P_0 is the process's rate at the low-end reference temperature T_0 . To incorporate the adverse effects of high temperatures on this process, the right-hand side of Equation (2.50) can be multiplied by a hyperbolic temperature term that approaches unity as temperature decreases well below T_1 , equals zero at T_1 , and becomes increasingly negative as temperature approaches the fish's upper thermal tolerance limit $T_L = T_2$. Modifying Equation (2.50) in this fashion yields

$$\frac{dP}{dT} = \varepsilon P \left(\frac{T - T_1}{T - T_2} \right) \quad (2.51)$$

whose solution is

$$P = P_0 \exp[\varepsilon(T - T_0)] \left(\frac{T_2 - T}{T_2 - T_0} \right)^{\varepsilon(T_2 - T_1)} \quad (2.52)$$

If one assumes, without loss of generality, that $T_0 = 0$, the preceding equation can be simplified to

$$P = P_0 \exp(\varepsilon T) \left(1 - \frac{T}{T_2} \right)^{\varepsilon(T_2 - T_1)} \quad (2.53)$$

Figure 2.3 displays the behavior of this equation for $P_0 = 1$ and $T_2 = 36$ Celsius as a function of ε and T_1 . Although these equations apparently have not been used to describe

physiological responses of fish, their utility for doing so is discussed subsequently in Section 3.3. For other applications of Equations (2.52) and (2.53) see Lassiter and Kearns (1974), Lassiter (1975), and Swartzman and Bentley (1979). Note that when $T_1 = T_2$, Equation (2.53) reduces to Equation (2.49).

2.6. Modeling Growth of Fish

Although the preceding algorithms for modeling chemical bioaccumulation in fish depend on a fish's live weight, BASS does not directly simulate the live weight of fish. Instead, it simulates the dry weight of fish as the mass balance of feeding, egestion, respiration, and excretion and then calculates the fish's associated wet weight using the following relationships

$$W_w = W_a + W_d = W_a + W_l + W_o \quad (2.54)$$

$$P_l = l_1 W_w^{l_2} \quad (2.55)$$

$$P_a = a_0 - a_1 P_l \quad (2.56)$$

$$P_o = 1 - P_a - P_l \quad (2.57)$$

where W_a , W_d , W_l , and W_o are the fish's aqueous, dry, lipid, and non-lipid organic weights, respectively; and a_0 , a_1 , l_1 , and l_2 are empirical constants. Whereas Equations (2.54) and (2.57) are simply assertions of mass conservation, Equations (2.55) and (2.56) are purely empirical functions. Although Equation (2.55) is assumed because simple power functions of this form adequately describe many morphometric relationships for most organisms, Equation (2.56) is based on the results of numerous field and laboratory studies (Eschmeyer and Phillips 1965, Brett et al. 1969, Groves 1970, Elliott 1976a, Staples and Nomura 1976, Craig 1977, Shubina and Rychagova 1981, Beamish and Legrow 1983, Weatherley and Gill 1983, Flath and Diana 1985, Lowe et al. 1985, Kunisaki et al. 1986, Morishita et al. 1987). These equations yield an expression for a fish's live weight that is a monotonically increasing, but nonlinear, function of the fish's dry weight.

BASS calculates a fish's realized feeding by first estimating its maximum *ad libitum* consumption (F_d^* g dry wt/d) and then adjusting this potential by the availability of appropriate prey as described in the next section. Because a variety of models are commonly used to describe the maximum feeding of fish, BASS is coded to allow users the option of using any one of four different feeding models for any particular age / size class of fish. The first formulation that can be used is a temperature-dependent power function

$$F_d^* = f_1 W_w^{f_2} \exp(f_3 T) \left(1 - \frac{T}{T_2} \right)^{f_3(T_2 - T_1)} \quad (2.58)$$

where f_1, f_2, f_3, T_1 , and T_2 are empirical constants specific to the fish's feeding.

A commonly used alternative to the preceding allometric model is the Rashevsky-Holling model that is defined by the equations

$$\begin{aligned} F_d^* &= \phi_{dd} (G_{\max} - G) \\ \frac{dG}{dt} &= F_d^* - A_d - E_d \end{aligned} \quad (2.59)$$

where ϕ_{dd} is the fish's *ad libitum* feeding rate (g dry wt/g dry wt/d); G_{\max} is the maximum amount of food (g dry wt/fish) that the fish's stomach / gut can hold; G is the actual amount of food (g dry wt/fish) present in the gut; and A_d and E_d are the fish's assimilation and egestion, respectively, in units of g dry wt/d (Rashevsky 1959, Holling 1966). Given a fish's gut capacity G_{\max} , feeding time t_{sat} to satiation, and satiating meal size M_{sat} , ϕ_{dd} can be estimated using the equations

$$F_d^*(t) = \int_0^t \phi_{dd} [G_{\max} - F_d^*(\tau)] d\tau \quad (2.60)$$

$$\frac{dF_d^*}{dt} = \phi_{dd} (G_{\max} - F_d^*) \quad (2.61)$$

$$-\phi_{dd} t_{sat} = \ln \left(1 - \frac{M_{sat}}{G_{\max}} \right) \quad (2.62)$$

where $F_d^*(t)$ is the total food consumed during the interval $(0, t)$ and $M_{sat} = F_d^*(t_{sat})$ (also see Dunbrack 1988). Alternatively, ϕ_{dd} can be estimated by simply assuming that $M_{sat} = 0.95 \times G_{\max}$ in which case

$$\phi_{dd} = -\frac{\ln 0.05}{t_{sat}} \quad (2.63)$$

For planktivores, BASS can also estimate a fish's maximum ingestion using the clearance volume model

$$F_d^* = \Psi Q_{cl} \quad (2.64)$$

where Ψ is the plankton standing stock (g dry wt/L); and Q_{cl} is the planktivore's clearance volume (L/d) that is assumed to be given by

$$Q_{cl} = q_1 W_w^{q_2} \exp(q_3 T) \left(1 - \frac{T}{T_2} \right)^{q_3 (T_2 - T_1)} \quad (2.65)$$

where q_1, q_2, q_3, T_1 , and T_2 are empirical constants specific to the fish's filtering rate.

The fourth and final feeding model back-calculates a fish's expected feeding based on knowing the fish's expected growth and routine respiratory demands. In particular, because assimilation, egestion, specific dynamic action, and excretion are assumed to be linear functions of feeding and routine respiration as discussed subsequently, it is a straightforward matter to calculate a fish's expected ingestion given its expected growth and respiration. When users elect this feeding option, BASS assumes that the fish's specific growth rate $\gamma = W_w^{-1} dW_w/dt$ (1/d) is given by

$$\gamma = g_1 W_w^{g_2} \exp(g_3 T) \left(1 - \frac{T}{T_2} \right)^{g_3 (T_2 - T_1)} \quad (2.66)$$

where g_1, g_2, g_3, T_1 , and T_2 are empirical constants specific to the fish's growth rate. See Thomann and Connolly (1984) for additional discussion of the use of this feeding model.

When BASS estimates a fish's feeding rate using Equations (2.58), (2.64), or (2.66), the fish's assimilation and egestion are estimated as simple fractions of its realized ingestion F_d , i.e.,

$$A_d = \alpha_f F_d \quad (2.67)$$

$$E_d = (1 - \alpha_f) F_d \quad (2.68)$$

where α_f is the fish's net food assimilation efficiency that is a weighted average of its assimilation efficiencies for invertebrate, piscine, and vegetative prey. However, when the Rashevsky-Holling feeding model is used, BASS calculates these fluxes by substituting F_d with a function that describes the fish's pattern of intestinal evacuation. The general form of this function is assumed to be

$$EV = e_1 G^{e_2} \exp(e_3 T) \left(1 - \frac{T}{T_2} \right)^{e_3 (T_2 - T_1)} \quad (2.69)$$

where e_1, e_2, e_3, T_1 , and T_2 are empirical constants specific to the fish's gastric evacuation.

The numerical value of this function's exponent, e_2 , depends both on characteristics of the food item being consumed and on the mechanisms that presumably control gastrointestinal motility and digestion (Jobling 1981, 1986, 1987). For example, when gut clearance is controlled by intestinal peristalsis, e_2 should be approximately equal $\frac{1}{2}$ since peristalsis is stimulated by circumferential pressure exerted by the intestinal contents that, in turn, is proportional to the square root of the contents mass. On the other hand, when surface area controls the rate of digestion, e_2 should be approximately either $\frac{2}{3}$ or unity. If the fish consumes a small number of large-sized prey (e.g., a piscivore), $e_2 = \frac{2}{3}$ may be the appropriate surface area model. On the other hand, if the fish consumes a large number of smaller,

relatively uniform-sized prey (e.g., a planktivore or drift feeder), $e_2 = 1$ is more appropriate since total surface area and total volume of prey become almost directly proportional to one another. When $e_2 = 1$, the Rashevsky-Holling model (i.e., Equation (2.59)) is analogous to the Elliott-Persson model for estimating daily rations of fish (Elliott and Persson 1978). Finally, Olson and Mullen (1986) outlined a process-based model that even suggests $e_2 = 0$.

A fish's specific dynamic action, i.e., the respiratory expenditure associated with the digestion and assimilation of food, is modeled as a constant fraction of the fish's assimilation. In particular,

$$SDA = \sigma A_d \quad (2.70)$$

where σ generally varies between 0.15 and 0.20 (Ware 1975, Tandler and Beamish 1981, Beamish and MacMahon 1988).

BASS assumes that body weight losses via metabolism are due entirely to the respiration of carbon dioxide and the excretion of ammonia. A fish's respiratory loss R is therefore calculated from its routine oxygen consumption, R_{O_2} (g O₂/d), using a respiratory quotient, RQ (L CO₂ respired)/L O₂ consumed), as follows

$$R = \frac{12 \text{ g C}}{\text{mole CO}_2} \cdot \frac{\text{mole CO}_2}{22.4 \text{ L CO}_2} \cdot RQ \cdot \frac{22.4 \text{ L O}_2}{\text{mole O}_2} \cdot \frac{\text{mole O}_2}{32 \text{ g O}_2} \cdot R_{O_2} = \frac{12}{32} \cdot RQ \cdot R_{O_2} \quad (2.71)$$

BASS calculates a fish's routine oxygen consumption as a constant multiple RB of its standard basal oxygen consumption (Ware 1975) that is assumed to be temperature-dependent power function. In particular,

$$R_{O_2} = RB b_1 W_w^{b_2} \exp(b_3 T) \left(1 - \frac{T}{T_2} \right)^{b_3 (T_2 - T_1)} \quad (2.72)$$

where b_1, b_2, b_3, T_1 , and T_2 are empirical constants specific to the fish's standard basal oxygen consumption. Although ammonia excretion could be modeled using an analogous function (Paulson 1980, du Preez and Cockcroft 1988a, b), BASS calculates this flux as a constant fraction of the fish's total respiration since excretion and oxygen consumption generally track one another. For example, ammonia excretion increases after feeding, as does oxygen consumption (Savitz 1969, Brett and Zala 1975, Gallagher et al. 1984). Likewise, conditions that inhibit the passive excretion of ammonia also depress carbon dioxide excretion (Wright et al. 1989). Assuming that fish maintain a constant nitrogen/carbon ratio NC (g N/g C), BASS estimates a fish's excretory loss in body weight as

$$EX = \varepsilon NC (R + SDA) \quad (2.73)$$

where $\varepsilon = 17/14$ is the ratio of the molecular weight of ammonia to that of nitrogen.

2.7. Modeling Predator-Prey Interactions

BASS is designed to simulate aquatic food webs in which each age class of a species can feed upon other fish species, benthos, incidental terrestrial insects, periphyton / attached algae, phytoplankton, and zooplankton. The realized feeding of any given age class of fish is determined by the estimated maximum feeding rate of individuals within the cohort, the cohort's population size, and the biomass of prey that is available to the cohort; the latter quantity is the sum of the current biomass of potential prey minus the biomass of potential prey expected to be consumed by other fish cohorts that are more efficient foragers / competitors. BASS ranks the competitive abilities of different cohorts using the following assumptions:

ASSUMPTION 1. The competitive abilities and efficiencies of benthivores and piscivores are positively correlated with their body sizes (Garman and Nielsen 1982, East and Magnan 1991). Two general empirical trends support this assumption. The first of these is the trend for the reactive distances, swimming speeds, and territory sizes of fish to be positively correlated with their body size (Minor and Crossman 1978, Breck and Gitter 1983, Wanzenböck and Schiemer 1989, Grant and Kramer 1990, Miller et al. 1992, Keeley and Grant 1995, Minns 1995). Given two differently sized predators of the same potential prey, these trends would suggest that the larger predator is more likely to encounter that prey than is the smaller. Having encountered the prey, the other general trend for prey handling times to be inversely correlated with body size (Werner 1974, Miller et al. 1992) suggests that the larger predator could dispatch intercepted prey and resume foraging more quickly than the smaller predator. Also see Post et al. (1999) and Railsback et al. (2002).

ASSUMPTION 2. Unlike benthivores and piscivores, the competitive abilities and efficiencies of planktivores are inversely related to their body size due to their relative morphologies (Lammens et al. 1985, Johnson and Vinyard 1987, Wu and Culver 1992, Persson and Hansson 1999). Consequently, "large" planktivores only have access to the leftovers of "small" planktivores.

BASS calculates the relative frequencies $\{..., d_i, ...\}$ of the prey consumed by a cohort using dietary electivities, i.e.,

$$e_i = \frac{d_i - f_i}{d_i + f_i} \quad (2.74)$$

where f_i is the relative availability of the i -th prey with respect to all other prey consumed by the cohort. One can easily verify that

the range of dietary electivities is $-1 < e_i < 1$. One can also verify that if the fish does not eat a potential food item i that $e_i = -1$. Similarly, if the fish consumes a potential prey item i in direct proportion to the prey's relative abundance, then $e_i = 0$. BASS actually allows users to specify a fish's diet as either a set of fixed dietary frequencies $\{\dots, \bar{d}_i, \dots\}$, a set of electivities $\{\dots, \bar{e}_i, \dots\}$, or a combination of fixed frequencies and electivities $\{\dots, \bar{d}_i, \dots, \bar{e}_j, \dots\}$. To calculate a cohort's realized dietary composition, however, BASS converts all user supplied fixed dietary frequencies into their equivalent electivities using the simulated relative abundances $\{\dots, f_i, \dots\}$ of the cohort's potential prey. These electivities are then combined with any user specified electivities to form a set of unadjusted electivities $\{\dots, \hat{e}_i, \dots\}$ that is subsequently converted into a consistent set of realized electivities $\{\dots, e_i, \dots\}$. Using these realized electivities, BASS then calculates the cohort's realized dietary frequencies using

$$d_i = \left(\frac{1 + e_i}{1 - e_i} \right) f_i \quad (2.75)$$

The important step in this computational process is the conversion of the unadjusted electivities into a set of realized electivities. Although this conversion is sometimes unnecessary, it is generally needed to insure that the sum of the dietary frequencies calculated by Equation (2.75) equals 1. One can verify that the condition that guarantees $\sum d_i = 1$ is

$$\sum_{i=1}^n \frac{f_i}{1 - e_i} = 1 \quad (2.76)$$

See Appendix C. When Equation (2.76) is not satisfied for a given set of electivities $\{\dots, \hat{e}_i, \dots\}$ and relative prey availabilities $\{\dots, f_i, \dots\}$, BASS transforms the given electivities using a linear transformation that maps $\hat{e}_i = -1$ into $e_i = -1$ and $\max(\dots, \hat{e}_i, \dots)$ into $e_i < 1$. The general form of this transformation is

$$e_i = m(\hat{e}_i + 1) - 1 \quad (2.77)$$

where $0 < m < 2 \max(\dots, \hat{e}_i, \dots)^{-1} + 1$. Besides insuring that $\sum d_i = 1$, this transformation also preserves the relative preferences represented in the original base set $\{\dots, \hat{e}_i, \dots\}$.

Because many studies have shown a strong positive correlation between the body sizes of piscivorous fish and the forage fish that they consume (Parsons 1971, Lewis et al. 1974, Timmons et

al. 1980, Gillen et al. 1981, Knight et al. 1984, Moore et al. 1985, Stiefvater and Malvestuto 1985, Storck 1986, Jude et al. 1987, Johnson et al. 1988, Yang and Livingston 1988, Brodeur 1991, Elrod and O'Gorman 1991, Hambright 1991, Juanes et al. 1993, Mattingly and Butler 1994, Hale 1996, Madenjian et al. 1998, Margenau et al. 1998, Mittelbach and Persson 1998, Bozek et al. 1999), when BASS uses the aforementioned procedure to calculate piscivorous interactions, only a specific size range of forage fish is assumed to be available to a piscivorous cohort. BASS characterizes this size spectrum by using either linear or complementary exponential functions to describe the maximum, minimum, and mode of prey body lengths that a predator of a given body length will ingest. In particular, these key features of a fish's prey spectrum are described by

$$L_{\max} = \begin{cases} \alpha_1 + \alpha_2 L_{\text{predator}} \\ \alpha_1 + \alpha_2 \exp(\alpha_3 L_{\text{predator}}) \end{cases} \quad (2.78)$$

$$L_{\min} = \begin{cases} \beta_1 + \beta_2 L_{\text{predator}} \\ \beta_1 + \beta_2 \exp(\beta_3 L_{\text{predator}}) \end{cases} \quad (2.79)$$

$$L_{\text{mode}} = \begin{cases} \gamma_1 + \gamma_2 L_{\text{predator}} \\ \gamma_1 + \gamma_2 \exp(\gamma_3 L_{\text{predator}}) \end{cases} \quad (2.80)$$

where α_1 , α_2 , and α_3 are empirical constants describing the fish's maximum length of prey; β_1 , β_2 , and β_3 are empirical constants describing the fish's minimal length of prey; and γ_1 , γ_2 , and γ_3 are empirical constants describing the mode of the length of prey ingested by the fish. The relative frequencies f_i of forage fish available to a piscivorous cohort are then calculated relative to the sum of forage fish biomasses whose body lengths are both greater than L_{\min} and less than L_{\max} minus the biomass of those prey sizes predicted to be consumed by more efficient piscivorous cohorts (see Assumption 1).

When two or more cohorts of a forage species i can be consumed by a piscivore, the relative frequencies of those cohorts s_{ij} in the piscivore's diet are calculated assuming that prey sizes follow a simple triangular distribution defined by Equations (2.78) through (2.80). For example, let L_{i1} and L_{i2} denote the body lengths of two age classes of species i that are prey for the cohort. If P_{ij} is the triangular distribution function

$$P_{ij} = \begin{cases} \frac{2(L_{ij} - L_{\min})}{(L_{\max} - L_{\min})(L_{\text{mode}} - L_{\min})} & \text{for } L_{ij} < L_{\text{mode}} \\ \frac{2(L_{\max} - L_{ij})}{(L_{\max} - L_{\min})(L_{\max} - L_{\text{mode}})} & \text{for } L_{ij} > L_{\text{mode}} \end{cases} \quad (2.81)$$

the relative frequencies of these two age classes in the cohort's diet are calculated to be $s_{i1} = d_i [P_{i1} / (P_{i1} + P_{i2})]$

and $s_{i2} = d_i [P_{i2}/(P_{i1} + P_{i2})]$. If only one age class of a forage species is vulnerable to the cohort, then $s_{ij} = d_i$.

If while calculating the dietary frequencies of a piscivorous cohort BASS predicts that the cohort's available prey is insufficient to satisfy its desired level of feeding, BASS reassigns the cohort's unadjusted electivities $\{\dots, \hat{e}_i, \dots\}$ in a manner to simulate prey switching. These reassignments are based on the following assumption:

ASSUMPTION 3. When forage fish become limiting, piscivores switch to benthic macroinvertebrates or incidental terrestrial insects as alternative prey. However, piscivores that must switch to benthos or that routinely consume benthos in addition to fish are less efficient benthivores than are obligate benthivores (Hanson and Leggett 1986, Lacasse and Magnan 1992, Bergman and Greenberg 1994). Consequently, only the leftovers of non-piscivorous benthivores are available to benthic feeding piscivores. If such resources are still insufficient to satisfy the piscivores' metabolic demands, piscivores are assumed then to switch to planktivory (Werner and Gilliam 1984, Magnan 1988, Bergman and Greenberg 1994). In this case, piscivores have access only to the leftovers of non-piscivorous planktivores. Using this assumption, BASS first assigns the cohort's electivity for benthos to zero regardless of its previous value. BASS also reassigns any other electivity that does not equal -1, to zero.

If benthos becomes limiting for benthivores, or if plankton becomes limiting for planktivores, BASS assumes that benthivores can shift their diets to include plankton and terrestrial insects and that planktivores can shift their diets to include benthos and terrestrial insects. See, for example, Ingram and Ziebell (1983).

After BASS has calculated a cohort's dietary composition, it then assigns the cohort's individual realized feeding rate adjusted for prey availability as

$$F_d = \max \left(F_d^*, N^{-1} \sum_{e_j^* = -1} AB_j \right) \quad (2.82)$$

where F_d^* is the cohort's maximum individual ingestion (g dry wt/fish); N is the cohort's population size (fish/ha), and AB_j is the biomass (g dry wt/ha) of prey j that is available to that cohort. Using its predicted dietary compositions and realized feeding rates, BASS then calculates the predatory mortalities for each fish cohort and nonfish compartments.

2.8. Modeling Dispersal, Non-Predatory Mortalities, and Recruitment

The algorithm that BASS employs to simulate a species' dispersal

and non-predatory mortality is based on the general empirical observation that population densities of most vertebrates can be adequately characterized by the self-thinning power function relationship

$$N = a W_w^{-b} \quad (2.83)$$

where N is the species' or cohort's density (fish/ha) and W_w is the species' or cohort's mean live body weight (Damuth 1981, Peters and Raelson 1984, Juanes 1986, Robinson and Redford 1986, Dickie et al. 1987, Boudreau and Dickie 1989, Gordoia and Duarte 1992, Randall et al. 1995, Dunham and Vinyard 1997, Steingrímsson and Grant 1999, Dunham et al. 2000, Guíñez 2005). For fish the body weight exponent b generally varies from 0.75 to 1.5 (Boudreau and Dickie 1989, Grant and Kramer 1990, Gordoia and Duarte 1992, Elliott 1993, Bohlin et al. 1994, Randall et al. 1995, Dunham and Vinyard 1997, Grant et al. 1998, Dunham et al. 2000, Knouft 2002, Keeley 2003). Larger exponents ranging from 1.5 to 3.0, however, have also been reported (Steingrímsson and Grant 1999). If Equation (2.83) is differentiated with respect to time, it immediately follows that a species' population dynamics can be modeled using the time-varying, linear differential equation

$$\frac{dN}{dt} = \frac{-a b W_w^{-b}}{W_w} \frac{dW_w}{dt} = -b \gamma N \quad (2.84)$$

where $\gamma = W_w^{-1} dW_w/dt$ is the species' specific growth rate. Consequently, $b\gamma$ corresponds to the cohort's total mortality rate. Readers interested in detailed discussions concerning the underlying process-based interpretation and general applicability of this result should consult Peterson and Wroblewski (1984), McGurk (1993, 1999), and Lorenzen (1996).

Because Equations (2.83) and (2.84) encompass the cohort's predatory mortality, non-predatory mortality, and dispersal, and because BASS separately models the cohort's predatory mortality, BASS assumes that the cohort's combined rate of dispersal (EM) and non-predatory mortality (NM) is simply a fraction δ of $b\gamma$. In particular,

$$EM + NM = \delta b \gamma N \quad (2.85)$$

If community population dynamics are strongly dominated by predation, the fraction δ will be "small" (e.g., $\delta < 0.5$) for forage fishes and "large" (e.g., $\delta > 0.5$) for predatory species. However, if community population dynamics are dominated by dispersal mechanisms related to competition for food, space, or other limiting community resource, the fraction δ will be large for both forage and predatory species alike.

BASS estimates a species' recruitment by assuming that each species turnover a fixed percentage of its potential spawning biomass into new young-of-year (YOY). This percentage is

referred to as the species' reproductive biomass investment (rbi). The species' spawning biomass is defined to be the total biomass of all cohorts whose body lengths are greater than or equal to a specified minimum value (tl_{r0}) marking the species' sexual maturation. When reproduction is simulated, the body weight of each sexually mature cohort is decreased by its rbi and the total number of YOY that are recruited into the population as a new cohort is estimated by simply dividing the species' total spawned biomass by the species' characteristic YOY body weight. Although this formulation does not address the myriad of factors known to influence population recruitment, it is logically consistent with the spawners' abundance model for fish recruitment. See Myers and Barrowman (1996) and Myers (1997).

2.9. Modeling Habitat Effects

Although BASS does not explicitly model physical habitat features of the fish community of concern, it does allow users to specify habitat suitability multipliers on the feeding, reproduction / recruitment, and dispersal / non-predatory mortality for any or all species. Because these multipliers are assumed to be analogous to subcomponents of habitat suitability indices, they are assumed to take values from 0 to 1. If these multipliers are not specified by the user, BASS assigns them the default value of 1.

When feeding habitat multipliers ($HSI_{feeding}$) are specified, BASS uses the specified parameters as simple linear multipliers on the fish's maximum rate of ingestion, i.e.,

$$F_d(\text{habitat}) = HSI_{feeding} F_d^* \quad (2.86)$$

The resulting adjusted maximum feeding rate then replaces F_d^* in Equation (2.82). These multipliers are assumed to modify the fish's ability to perceive or to intercept prey either by effecting the fish's reactive distance, foraging patterns, etc. or by providing modified refuges for its potential prey. Habitat interactions that actually change the abundance of potential prey should not be specified as feeding habitat multipliers since these interactions are automatically addressed by the algorithms outlined in Section 2.7.

Like the aforementioned feeding habitat multipliers, BASS uses any specified recruitment habitat multipliers ($HSI_{recruitment}$) as simple linear multipliers on the number of young-of-year that is recruited into the species population, i.e.,

$$N_0(\text{habitat}) = HSI_{recruitment} N_0 \quad (2.87)$$

These multipliers can represent either the availability of suitable spawning sites or the ability of the otherwise successful spawns to result in the expected numbers of young-of-year as discussed

in Section 2.8.

Finally, when habitat multipliers ($HSI_{survival}$) are specified for dispersal / non-predatory mortality, the specified values are assumed to dynamically control the species self-thinning exponent b (see Section 2.8) such that the exponent is maximum for $HSI_{survival} = 0$ and minimum for $HSI_{survival} = 1$. Thus, as habitat suitability decreases, dispersal and non-predatory mortality increases, and vice versa (see Equation (2.85)). In between this range the species self-thinning exponent is assumed to respond linearly to changing $HSI_{survival}$, i.e.,

$$b(\text{habitat}) = (1 - HSI_{survival})(b_{max} - b_{min}) + b_{min} \quad (2.88)$$

Because constructing the aforementioned habitat suitability multipliers in a general or standard way is not a trivial issue, BASS relegates their construction to the user. Nevertheless, users might consider several obvious starting points when simulating habitat effects using BASS. Turbidity, for example, is known to affect the foraging abilities of both prey and predatory fishes, and one could readily use results of published studies (e.g., Vandenberg et al. 1991, Barrett 1992, Gregory 1993, Gregory and Northcote 1993, Miner and Stein 1996, Reid et al. 1999, Vogel and Beauchamp 1999, Bonner and Wilde 2002, de Robertis et al. 2003, Sweka and Hartman 2003) to estimate feeding multipliers for Equation (2.86) as power functions or polynomials of turbidity. Field-based HSI's are often estimated by logistic regression of presence-absence data without specifying the underlying mechanisms that actually determine habitat suitability for a species. Such HSI's could be used as habitat multipliers for species' recruitment (Equation (2.87)) or persistence/survival (Equations (2.85) and (2.88)) depending on the user's own interpretation of what the indices most likely represent.

2.10. Modeling Nonfish Compartments

BASS assumes that the nonfish components of a community of concern can be treated as four lumped compartments, i.e., benthos, periphyton/attached algae, phytoplankton, and zooplankton. These compartments can be treated either as community forcing functions or as bona fide state variables. In the later case, the required compartmental dynamics are simulated using the simple mass balance model

$$\frac{dY}{dt} = IP - R - \hat{F} - M \quad (2.89)$$

where Y is the compartment's biomass (g dry wt/m²); and the fluxes g dry wt/m²/d IP , R , \hat{F} , and M are the compartment's ingestion or photosynthesis, respiration, mortality due to fish consumption, and non-consumptive mortality and dispersal,

respectively. Except for \hat{F} , each of these fluxes is modeled as a linear function of the compartment's biomass, i.e.,

$$IP = \phi_{dd} Y \quad (2.90)$$

$$R = \rho Y \quad (2.91)$$

$$M = \mu Y \quad (2.92)$$

where the rate coefficients (g dry wt/g dry wt/d) ϕ_{dd} , ρ , and μ are minimally functions of temperature and time.

For benthos, phytoplankton, and zooplankton that can be conceptualized as populations of organisms possessing similar body sizes, the rate coefficients ϕ_{dd} , ρ , and μ are estimated using temperature-dependent allometric relationships that describe the pertinent processes for the individuals comprising the compartment of interest. For example, consider the following formulation of benthos consumption. Assuming that \bar{W}_d is the average dry weight of individuals comprising the benthos compartment, it follows that the expected density of individuals within the benthos compartment is simply

$$N = \frac{Y}{\bar{W}_d} \quad (2.93)$$

Given that the consumption (g dry wt/d) of the aforementioned average benthic invertebrate is described by

$$C = c_1 \bar{W}_d^{c_2} \exp(c_3 T) \left(1 - \frac{T}{T_2}\right)^{c_3(T_2 - T_1)} \quad (2.94)$$

it follows that the ingestion of the benthos compartment at large can be modeled as

$$IP = C N$$

$$\begin{aligned} &= \left[c_1 \bar{W}_d^{c_2} \exp(c_3 T) \left(1 - \frac{T}{T_2}\right)^{c_3(T_2 - T_1)} \right] \frac{Y}{\bar{W}_d} \\ &= \left[c_1 \bar{W}_d^{c_2 - 1} \exp(c_3 T) \left(1 - \frac{T}{T_2}\right)^{c_3(T_2 - T_1)} \right] Y \\ &= \phi_{dd} Y \end{aligned} \quad (2.95)$$

Formulating compartmental ingestion, photosynthesis, and respiration using this method not only delineates an objective procedure to parameterize BASS, but also yields production relationships that are consistent with results reported by Plante and Downing (1989), Stockwell and Johannsson (1997), and Kuns and Sprules (2000). When estimating ϕ_{dd} , ρ , and μ for benthos, phytoplankton, or zooplankton using this approach,

BASS assumes that

$$P = p_1 \bar{W}_d^{p_2} \exp(p_3 T) \left(1 - \frac{T}{T_2}\right)^{p_3(T_2 - T_1)} \quad (2.96)$$

where P is the individual's ingestion, photosynthesis, or respiration in units of g dry wt/d; and p_1 , p_2 , p_3 , T_1 , and T_2 are empirical constants specific to the process of interest. Although BASS does not attempt to simulate the average individual body sizes of benthos, phytoplankton, or zooplankton, BASS does allow the user to vary these parameters as functions of time.

Because periphyton communities are typically complex amalgamations of filamentous and unicellular algae, it is difficult to conceptualize this compartment as a population of archetypical individuals and to employ the preceding model parameterization scheme. Consequently, for periphyton BASS assumes that ϕ_{dd} , ρ , and μ are generally temperature-dependent allometric functions of the compartment's biomass, i.e.,

$$\phi_{dd} = \alpha_1 Y^{\alpha_2} \exp(\alpha_3 T) \left(1 - \frac{T}{T_2}\right)^{\alpha_3(T_2 - T_1)} \quad (2.97)$$

$$\rho = \beta_1 Y^{\beta_2} \exp(\beta_3 T) \left(1 - \frac{T}{T_2}\right)^{\beta_3(T_2 - T_1)} \quad (2.98)$$

$$\mu = \delta_1 Y^{\delta_2} \exp(\delta_3 T) \left(1 - \frac{T}{T_2}\right)^{\delta_3(T_2 - T_1)} \quad (2.99)$$

The rationale for these formulations is based on the assumption that the primary production, respiration, and possibly mortality of periphyton communities are generally limited by their surface-volume relationships that are implicitly represented by these equations.

Because ϕ_{dd} is generally much greater than ρ , the astute reader will recognize that Equations (2.89) - (2.92) will predict unbounded autocatalytic growth for any nonfish compartment whose predatory mortality and non-predatory mortality/dispersal does not precisely balance its intrinsic growth rate. To prevent such unrealistic dynamics, BASS internally estimates a physiologically based carrying capacity for each nonfish compartment based on its projected daily oxygen consumption and the community's prevailing dissolved oxygen content. In particular, BASS assumes that compartmental oxygen consumption cannot exceed the dissolved oxygen content corresponding to the difference between the community's prevailing dissolved oxygen concentration (DOC) and an assumed hypoxic threshold of 4 mg O₂ /L. When the compartment's daily oxygen consumption is predicted to exceed this available dissolved oxygen content, compartmental growth

is suspended by equating the compartment's feeding/photosynthesis to its projected respiration.

BASS assumes that the rates of chemical bioaccumulation in nonfish compartments are rapid enough to enable chemical concentrations within these components to be calculated using simple bioaccumulation factors. In particular,

$$C_{nf} = BAF_{nf} C_w \quad (2.100)$$

where C_{nf} is the chemical concentration (g/g dry wt) in the compartment of concern. BASS enables users to specify the bioaccumulation factor BAF_{nf} (ml/g dry wt) for Equation (2.100) either as an empirically derived constant, as a quantitative structure activity relationship (QSAR), or as the ratio of the chemical's uptake rate to the sum of its excretion rate and the compartment's growth rate. When BAF_{nf} is specified as a QSAR, BASS assumes that

$$BAF_{nf} = b_1 K_{ow}^{b_2} \quad (2.101)$$

where b_1 and b_2 are empirical constants. When BAF_{nf} is specified by the compartment's chemical exchange rates and growth rate, BASS assumes that

$$BAF_{nf} = \frac{k_1}{k_2 + \gamma} = \frac{k_1}{k_1/K_{nf} + \gamma} \quad (2.102)$$

where k_1 , k_2 , and γ are the rates of uptake, excretion, and growth, respectively, by individuals comprising the compartment; and K_{nf} is the compartment's thermodynamic bioconcentration factor that is defined analogously to Equation (2.7). For heterotrophs, Equation (2.102) assumes that direct chemical uptake and excretion with the ambient water are dominant over the dietary uptake and fecal excretion of the organisms of concern. Although this assumption is not satisfied for all benthic or planktonic heterotrophs, it does bypass the need to specify feeding rates, assimilation efficiencies, and dietary compositions for compartments that are actually mixed functional groups. For further discussions of Equation (2.102) and its generalization, readers should consult Connolly and Pedersen (1988), Thomann (1989), and Arnot and Gobas (2004).

2.11. Modeling Toxicological Effects

Narcosis is defined to be any reversible decrease in physiological function that is induced by chemical agents. Because the potency of narcotic agents was originally found to be correlated with their olive oil / water partition coefficients (Meyer 1899, Overton 1901), it was long believed that the principal mechanism of narcosis was the disruption of the transport functions of the lipid bilayers of biomembranes (Mullins 1954, Miller et al. 1973, Haydon et al. 1977, Janoff et al. 1981, Pringle et al. 1981). More recently, however, it has been acknowledged that narcotic

chemicals also partition into other macromolecular components besides the lipid bilayers of membranes. It is now widely accepted that partitioning of narcotic agents into hydrophobic regions of proteins and enzymes inhibit their physiological function either by changing their conformational structure or by changing the configuration or availability of their active sites (Eyring et al. 1973, Adey et al. 1976, Middleton and Smith 1976, Richards et al. 1978, Franks and Lieb 1982, 1984, Law et al. 1985, Lassiter 1990). In either case, however, the idea that the presence of narcotic chemicals increases the physical dimensions of various physiological targets to some "critical volume" that renders them inactive is fundamental (Abernethy et al. 1988). Consequently, narcotic chemicals can be treated as generalized physiological toxicants, and narcosis itself can be considered to represent baseline chemical toxicity for organisms. Although any particular chemical can act by a more specific mode of action under acute or chronic exposure conditions, all organic chemicals can be assumed to act minimally as narcotics (Ferguson 1939, McCarty and Mackay 1993).

Studies have shown that for narcotic chemicals there is a relatively constant chemical activity within exposed organisms associated with any given level of biological activity (Ferguson 1939, Brink and Posternak 1948, Veith et al. 1983). This relationship holds true not only for exposures to a single chemical but also for exposures to chemical mixtures. In the case of a mixture of chemicals, the sum of the chemical activities for each component chemical is constant for a given level of biological activity. Because narcotic chemicals can be treated as generalized physiological toxicants as already noted, it should not be too surprising that the effects of mixtures of chemicals possessing diverse specific modes of action not only often resemble narcosis but also appear to be additive in terms of their toxic effects (Barber et al. 1987, McCarty and Mackay 1993). For example, although most pesticides possess a specific mode of action during acute exposures, the joint action of pesticides is often additive and resembles narcosis (Hermanutz et al. 1985, Matthiessen et al. 1988, Bailey et al. 1997).

BASS simulates acute and chronic mortality assuming that the chemicals of concern are an additive mixture of narcotics. Because this assumption is the least conservative assumption that one could make concerning the onset of effects, mortalities predicted by BASS should signal immediate concern. When the total chemical activity of a fish's aqueous phase exceeds its calculated lethal threshold, BASS assumes that the fish dies and then eliminates that fish's age class from further consideration. The total chemical activity of a fish's aqueous phase is simply the sum of the fish's aqueous phase chemical activity for each chemical. BASS calculates the aqueous phase chemical activity of each chemical using the following formulae

$$A_a = \gamma_a M_a$$

$$M_a = \frac{C_a}{10^3 MW} = \frac{C_f}{10^3 MW K_f} \quad (2.103)$$

where A_a is the chemical's aqueous activity; γ_a is the chemical's aqueous activity coefficient (L/mol), the reciprocal of its sub-cooled liquid solubility; M_a is the chemical's molarity within the aqueous phase of the fish; and MW is the chemical's molecular weight (g/mol).

BASS estimates the lethal chemical activity threshold for each species as the geometric mean of the species' LA_{50} , i.e., the ambient aqueous chemical activity that causes 50% mortality in an exposed population. These lethal thresholds are calculated using the above formulae with user-specified LC_{50} 's substituted for C_a . These calculations are based on two important assumptions. The first assumption is that the exposure time associated with the specified LC_{50} is sufficient to allow almost complete chemical equilibration between the fish and the water. The second assumption is that the specified LC_{50} is the minimum LC_{50} that kills the fish during the associated exposure interval. Fortunately, most reliable LC_{50} 's satisfy these two assumptions. See Lassiter and Hallam (1990) for a comprehensive model-

based analysis of these issues.

Three points should be mentioned regarding the above approach to modeling ecotoxicological effects. First, for narcotic chemicals this approach is analogous to the toxic unit approach for evaluating the toxicity of mixtures (Calamari and Alabaster 1980, K onemann 1981a, b, Hermens and Leeuwangh 1982, Hermens et al. 1984a, Hermens et al. 1984b, Broderius and Kahl 1985, Hermens et al. 1985b, Hermens et al. 1985c, Hermens et al. 1985a, Dawson 1994, Peterson 1994). Second, the approach is also analogous to the critical body residue (CBR) and total molar body residue (TBR) approaches proposed by McCarty and Mackay (1993), Verhaar et al. (1995), and van Loon et al. (1997). Lastly, although sublethal effects are not presently modeled by BASS, BASS's simulation results can be used to indicate when sublethal effects that are induced by narcotic agents would be expected to occur. Results reported by Hermens et al. (1984b) indicate that for *Daphnia* the ratio of the EC_{50} for reproductive impairment to the LC_{50} is generally on the order of 0.15 - 0.30 for chemicals whose $\log K_{ow}$ range from 4 to 8. For individual growth inhibition, however, the mean EC_{50} to LC_{50} ratio for *Daphnia* in 16 day chronic exposures was approximately 0.77 (Hermens et al. 1984a, Hermens et al. 1985b). Also see Roex et al. (2000).

Table 2.1 Summary of the notation used for model development excluding empirical parameters describing fundamental model processes, rates, or rate coefficients.

A_a	chemical activity in aqueous fraction of the fish (dimensionless)
A_d	assimilation rate (g dry wt/d)
B_f	chemical burden in whole fish ($\mu\text{g}/\text{fish}$)
BAF_{nf}	bioaccumulation factor for nonfish prey (ml/g dry wt)
C_a	chemical concentration in aqueous fraction of the fish ($\mu\text{g}/\text{ml}$)
C_{ae}	chemical concentration in aqueous fraction of intestinal contents ($\mu\text{g}/\text{ml}$)
C_B	chemical concentration in bulk interlamellar water ($\mu\text{g}/\text{ml}$)
C_e	chemical concentration in egesta/feces ($\mu\text{g}/\text{ml}$)
C_f	chemical concentration in whole fish ($\mu\text{g}/\text{g}$ wet wt)
C_{de}	chemical concentration in dry organic fraction of intestinal contents ($\mu\text{g}/\text{g}$ dry wt)
C_l	chemical concentration in lipid ($\mu\text{g}/\text{g}$ dry wt)
C_{nf}	chemical concentration nonfish prey ($\mu\text{g}/\text{g}$ dry wt)
C_o	chemical concentration in non lipid organic matter ($\mu\text{g}/\text{g}$ dry wt)
C_p	chemical concentration in prey ($\mu\text{g}/\text{g}$ wet wt)
C_w	chemical concentration in environmental water ($\mu\text{g}/\text{ml}$)
C_{w,O_2}	oxygen concentration in environmental water ($\mu\text{g}/\text{ml}$)
d	interlamellar distance (cm)
d_i	the relative frequency of prey i in a fish's diet (dimensionless)
D	aqueous diffusion coefficient (cm^2/s)
e_i	the electivity prey i in a fish's diet (dimensionless)
E_d	egestive rate (g dry wt/d)
E_w	egestive rate (g wet wt/d)
EM	emigration/dispersal (fish/ha/d)
EX	excretory rate (g dry wt/d)
f_i	the relative frequency of prey i in the field (dimensionless)
F_d^*	maximum feeding rate (g dry wt/d)
F_d	feeding rate (g dry wt/d)
F_w	feeding rate (g wet wt/d)
G	mass of gut contents (g dry wt/fish)
h	height of secondary lamellae (cm)
$HSI_{feeding}$	habitat suitability index for cohort feeding (dimensionless)
$HSI_{recruitment}$	habitat suitability index for YOY recruitment (dimensionless)
$HSI_{survival}$	habitat suitability index for cohort survival (dimensionless)
J_{bt}	biotransformation of chemical ($\mu\text{g}/\text{s}$)
J_g	net chemical exchange across the gills ($\mu\text{g}/\text{s}$)
J_i	net chemical exchange across the intestine ($\mu\text{g}/\text{s}$)
k_2	apparent elimination rate coefficient (ml/g wet wt/d, g wet wt/g wet wt/d, or 1/d), i.e., $k_2 = (\gamma + k_{bt} + k_{ex})$
k_{bt}	chemical biotransformation rate coefficient (ml/g wet wt/d, g wet wt/g wet wt/d, or 1/d)
k_{ex}	chemical excretion rate coefficient (ml/g wet wt/d, g wet wt/g wet wt/d, or 1/d)
k_g	overall chemical conductance across the gill from the interlamellar water to the aqueous blood (cm/s)
k_m	chemical conductance through the gill membrane (cm/s)
K_e	partition coefficient for fecal matter (ml/g wet wt)
K_f	thermodynamic bioconcentration factor (ml/g wet wt)
K_l	partition coefficient between generic lipid and water (ml/g dry wt)
K_o	partition coefficient between non-lipid organic matter and water (ml/g dry wt)
K_{oc}	partition coefficient between organic carbon and water (ml/g dry wt)
K_{ow}	partition coefficient between n-octanol and water (ml/ml)
l	lamellar length (cm)
L	fish's body length (cm)

M_a	chemical molarity in aqueous fraction of the fish (mol/L)
N	population density (fish/ha)
N_{Gz}	Graetz number (dimensionless) = $(l D)/(V r^2)$
N_{Sh}	Sherwood number (dimensionless) = $(k_m r)/D$
NM	non-predatory mortality (fish/ha/d)
P_a	aqueous or moisture fraction of whole fish (g water/g wet wt = ml/g wet wt)
P_{ae}	aqueous or moisture fraction of feces/egesta (g water/g wet wt = ml/g wet wt)
P_{ap}	aqueous or moisture fraction of prey/food (g water/g wet wt = ml/g wet wt)
P_{de}	dry fraction of feces/egesta (g dry wt/g wet wt)
P_{dp}	dry fraction of prey/food (g dry wt/g wet wt)
P_d	dry fraction of whole fish (g dry wt/g wet wt), i.e., $P_d = (1 - P_a) = (P_l + P_o)$
P_l	lipid fraction of whole fish (g dry wt/g wet wt)
P_o	non-lipid organic fraction of whole fish (g dry wt/g wet wt)
PM	predatory mortality (fish/ha/d)
Q	ventilation volume (cm ³ /s)
r	hydraulic radius of interlamellar channels (cm), i.e., $r = 0.5 d$
R	routine respiratory rate (g dry wt/d)
R_{O_2}	oxygen consumption rate (mg O ₂ /s or g O ₂ /d)
SDA	specific dynamic action (g dry wt/d)
S_g	total gill surface area (cm ²)
T	temperature (Celsius)
V	average velocity of interlamellar flow (cm/s)
W_a	weight/volume of fish's aqueous phase (g water/fish or ml/fish)
W_d	weight of fish (g dry wt/fish)
W_l	weight of fish's lipid phase (g dry wt/fish)
W_o	weight of fish's nonlipid organic phase (g dry wt/fish)
W_w	weight of fish (g wet wt/fish)
X_g	cross sectional pore area of the gill (cm ²)
α_c	assimilation efficiency of chemical (dimensionless)
α_f	assimilation efficiency of food (g dry wt assimilated/g dry wt ingested)
α_{O_2}	oxygen assimilation efficiency of the gill (dimensionless)
γ	specific growth rate (g wet wt/g wet wt/d), i.e., $\gamma = W_w^{-1} dW_w/dt$
γ_a	chemical aqueous phase activity coefficient (L/mol)
ϵ_a	aqueous phase biotransformation rate coefficient (1/d)
Φ_{dd}	specific feeding rate (g dry wt/g dry wt/d)
Φ_{ww}	specific feeding rate (g wet wt/g wet wt/d)
η	solution viscosity (poise)
v	molar volume (cm ³ /mol)
ρ	lamellar density (lamellae/mm)

Figure 2.1 First eigenvalue and bulk mixing cup coefficient for Equation (2.28) as a function of gill Sherwood number and ventilation / perfusion ratio.

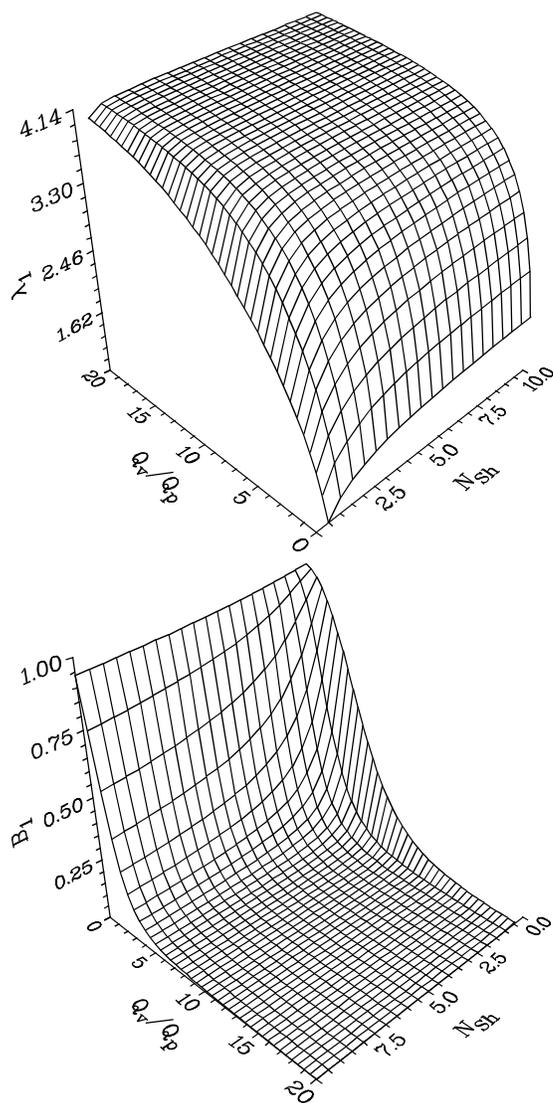


Figure 2.2 Second eigenvalue and bulk mixing cup coefficient for Equation (2.28) as a function of gill Sherwood number and ventilation / perfusion ratio.

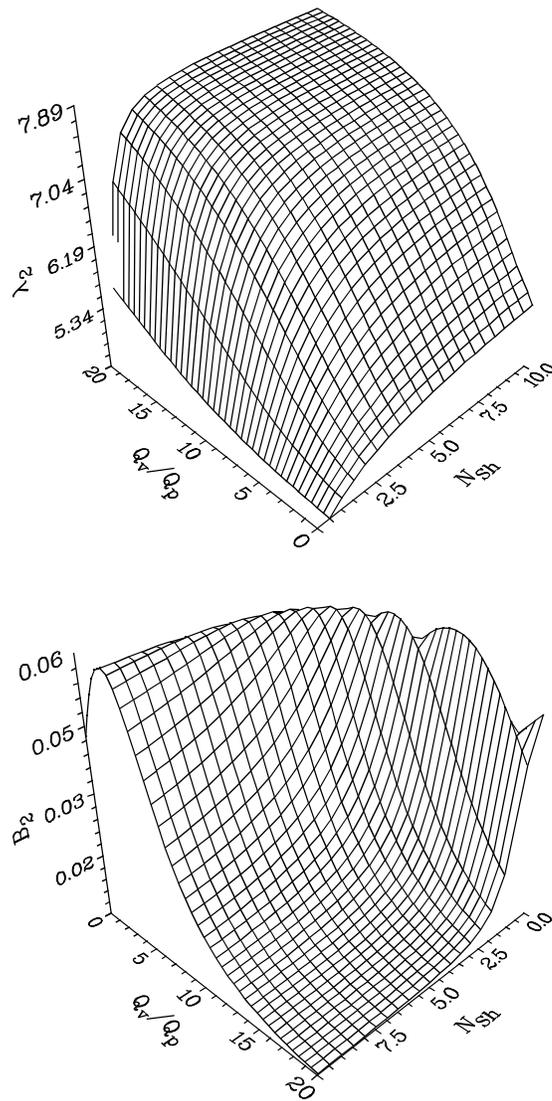
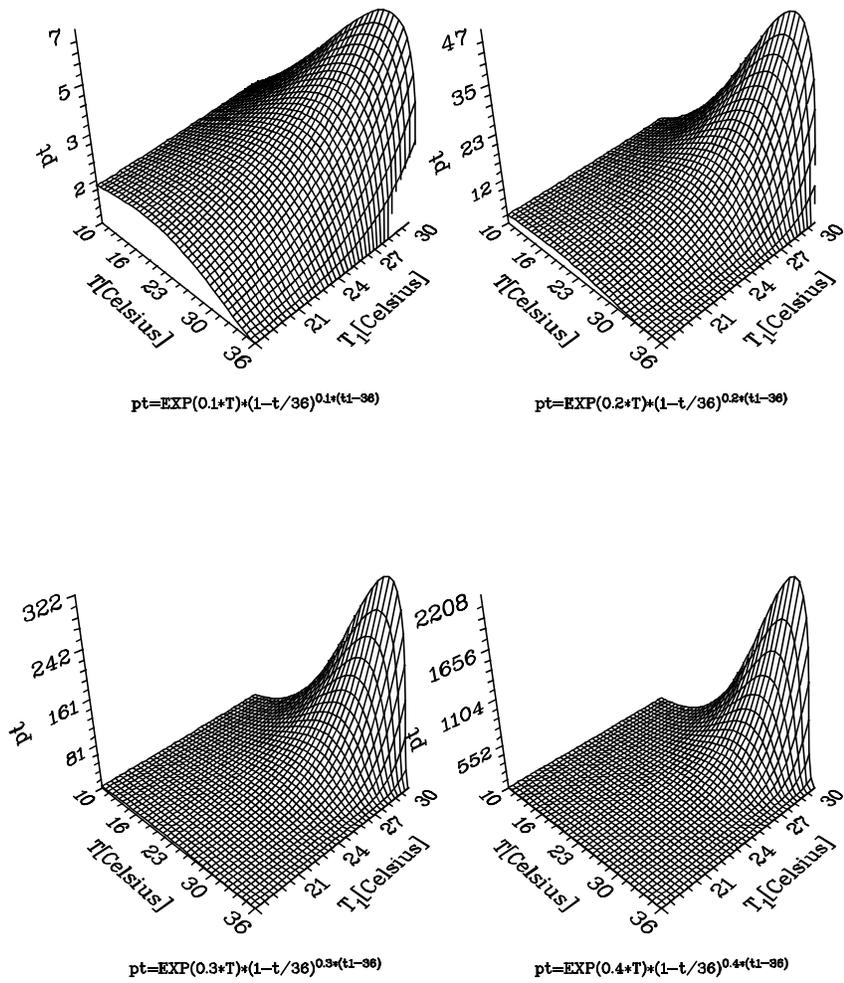


Figure 2.3 Functional behavior of Equation (2.53)



3. Model Parameterization

Because reliable application of a model depends not only on the validity of its formulation but also on its parameterization, important aspects regarding the parameterization of BASS's bioaccumulation and physiological algorithms are discussed below.

3.1. Parameterizing K_f

Superficially, estimation of a fish's thermodynamic bioconcentration factor K_f via Equation (2.7) appears to require a great deal of information. This task, however, is much simpler than it first appears. For example, given a fish's lipid fraction (see Equation (2.56)), it is a straightforward matter to calculate the fish's aqueous fraction using Equation (2.55). Having done so, one can then immediately calculate the fish's non-lipid organic fraction since P_o , P_l , and P_w must sum to unity (i.e., Equation (2.57)).

For an organic chemical, the partition coefficients K_l and K_o can be estimated using the chemical's octanol / water partition coefficient K_{ow} . Although triglycerides are the principal storage lipids of fish and it would seem reasonable to estimate K_l using a triglyceride / water partition coefficient, BASS assumes that K_l equals K_{ow} . To estimate K_o , BASS assumes that a fish's non-lipid organic matter is equivalent to organic carbon and uses Karickhoff's (1981) regression between the organic carbon / water partition coefficient (K_{oc}) and K_{ow} to estimate this parameter. Specifically,

$$K_o = K_{oc} = 0.411 K_{ow} \quad (3.1)$$

For metals or metallo-organic compounds such as methylmercury, the chemical's lipid partition coefficient K_l can again be assumed to equal its octanol / water partition coefficient K_{ow} . A metal's distribution coefficient into non-lipid organic matter, however, cannot be estimated using the K_{oc} relationship of Equation (3.1). For example, whereas the K_{ow} of methylmercury at physiological pH's is approximately 0.4 (Major et al. 1991), its distribution coefficient into environmental organic matter is on the order of 10^4 - 10^6 (Benoit et al. 1999b, Benoit et al. 1999a). O'Loughlin et al. (2000) report similar differences for organotin compounds. Whereas distribution coefficients for metals into fecal matter generally should be assigned values comparable to those used to model the environmental fate and transport of metals, distribution coefficients for metals into the non-lipid organic matter of fish should be assigned values 10 to 100 times higher to reflect the increased number and availability of sulfhydryl binding sites.

3.2. Parameters for Gill Exchange

To parameterize the gill exchange model, the fish's total gill area (S_g cm²), mean interlamellar distance (d cm), and mean lamellar length (l cm) must be specified. Each of these morphological variables is generally assumed to be an allometric power function of the fish's body weight, i.e.,

$$S_g = s_1 W_w^{s_2} \quad (3.2)$$

$$d = d_1 W_w^{d_2} \quad (3.3)$$

$$l = l_1 W_w^{l_2} \quad (3.4)$$

Although many authors have reported allometric coefficients and exponents for total gill surface area, coefficients and exponents for the latter two parameters are seldom available. Parameters for fish's mean interlamellar distance, however, can be estimated if the allometric function for the density of lamellae on the gill filaments, ρ (number of lamellae per mm of gill filament), i.e.,

$$\rho = \rho_1 W_w^{\rho_2} \quad (3.5)$$

is known. Fortunately, lamellar densities, like total gill areas, are generally available in the literature. See Barber (2003). BASS estimates d_1 and d_2 from ρ_1 and ρ_2 using the interspecies regression ($n = 28$, $r = -0.92$)

$$d = 0.118 \rho^{-1.19} \quad (3.6)$$

To overcome the scarcity of published morphometrics relationships for lamellar lengths, BASS uses the default interspecific regression ($n = 90$, $r = 0.92$)

$$l = 0.0188 W_w^{0.294} \quad (3.7)$$

Both of the preceding regressions are functional regressions rather than simple linear regressions (Rayner 1985, Jensen 1986); the data used for their development are taken from Saunders (1962), Hughes (1966), Steen and Berg (1966), Muir and Brown (1971), Umezawa and Watanabe (1973), Galis and Barel (1980), and Hughes et al. (1986).

To calculate lamellar Graetz and Sherwood numbers, BASS estimates a chemical's aqueous diffusivity (cm²/s), using the empirical relationship,

$$D = 2.101 \times 10^{-7} \eta^{-1.4} \nu^{-0.589} \quad (3.8)$$

where η is the viscosity (poise) of water; and ν is the chemical's molar volume (cm³/mol) (Hayduk and Laudie

1974). The diffusivity of a chemical through the gill membrane that is needed to estimate the membrane's permeability k_m is then assumed to equal one half the chemical's aqueous diffusivity (Piiper et al. 1986, Barber et al. 1988, Erickson and McKim 1990). The other quantity needed to estimate k_m is the thickness of the gill's epithelial layer. Although previous versions of BASS assumed a constant water-blood barrier thickness (β_e) equal to 0.0029 cm for all fish species, BASS now uses the interspecies allometric relationship

$$\beta_e = 9.17 \times 10^{-5} W_w^{0.261} \quad (3.9)$$

to estimate this parameter (Barber 2003).

To calculate ventilation / perfusion ratios BASS estimates the ventilation volumes (ml/hr) of fish from their oxygen consumption rates assuming an extraction efficiency of 60% and a saturated dissolved oxygen concentration (see Equation (2.12)). Perfusion rates (ml/hr) are estimated using

$$Q_p = (0.23 T - 0.78) 1.862 W_w^{0.9} \quad (3.10)$$

as the default for all species. Although this expression, in units of L/kg/hr, was developed by Erickson and McKim (1990) for rainbow trout (*Oncorhynchus mykiss*), it has been successfully applied to other fish species (Erickson and McKim 1990, Lien and McKim 1993, Lien et al. 1994).

The eigenvalues and bulk mixing cup coefficients needed to parameterize Equation (2.28) are interpolated internally by BASS from matrices of tabulated eigenvalues and mixing cup coefficients that encompass the range of Sherwood numbers (i.e., $1 < N_{Sh} < 10$) and ventilation / perfusion ratios (i.e., $1 < Q_v / Q_p < 20$) that are typical for fish (Hanson and Johansen 1970, Barron 1990, McKim et al. 1994, Sijm et al. 1994). See **Figure 2.1** and **Figure 2.2** of the previous chapter.

3.3. Bioenergetic and Growth Parameters

Parameterization of the physiological processes used by BASS to simulate fish growth generally poses no special problems since the literature abounds with studies that can be used for this purpose. The BASS Data Supplement summarizes literature data that have analyzed to date for use by BASS.

3.4. Procedures Used to Generate the BASS Database

BASS's database for fish ecological, morphological, and physiological parameters is generated by its own Fortran 95 software program. This program not only decodes functional expressions for BASS model parameters that have been reported in the literature but also calculates its own regressions using

data reported in the literature. Each species within the BASS database is assigned its own data file whose name corresponds to its genus and species. Thus, all literature data and regressions pertaining to largemouth bass are compiled into the BASS database file *micropterus_salmoides.dat*. Literature regressions are entered into BASS database files using the functional syntax outlined in chapter sections 4.3.3, 4.4.1, and 4.4.2 herein. Except for this syntax, all literature regressions are recorded as reported; any required units conversions are performed by the BASS database generator.

When literature regressions are not equivalent to the functional forms used by BASS, and their associated primary data are not reported, synthetic datasets are generated to estimate the needed parameters. For example, when a fish's oxygen consumption does not exhibit a temperature optimum, BASS assumes that this parameter is given by

$$so[mg(o2)/hr] = a W[g]^b \exp(c * t[celsius]) \quad (3.11)$$

or equivalently

$$\log so[mg(o2)/hr] = \log a + b \log W[g] + c t[celsius] \quad (3.12)$$

Although many researchers use similar expressions to report a fish's oxygen consumption, some researchers use the function

$$so[mg(o2)/hr] = a W[g]^b t[celsius]^c \quad (3.13)$$

or equivalently

$$\log so[mg(o2)/hr] = \log a + b \log W[g] + c \log t[celsius] \quad (3.14)$$

When such power functions of temperature are encountered, synthetic datasets of "observed/predicted" oxygen consumption are generated using the reported regressions for the reported range of body weights and temperatures. These synthetic data are then refitted to Equation (3.11).

A similar procedure for generating synthetic datasets is used to convert the temperature-dependent functions (Kitchell et al. 1977, Thornton and Lessem 1978) employed by the widely used Wisconsin Bioenergetics Fish Model (Hanson et al. 1997) into the hyperbolic Arrhenius formulation assumed by BASS.

Although the BASS database generator performs most of its parameter estimations using univariate statistics or ordinary linear least-squares regression analysis as appropriate, nonlinear least-squares regression analysis is used to estimate specific growth rates and physiological functions that are to be fitted to BASS's hyperbolic Arrhenius formulation. In these latter instances, BASS's database generator uses the NL2SOL Fortran 90 software that solves nonlinear least-squares

problems using a modified Newton's method with analytic Jacobians and a secant updating algorithm to compute the required Hessian matrix. See Dennis et al. (1981).

Estimation of Specific Growth Rates

BASS uses specific growth rates ($\gamma = W^{-1} dW/dt$) not only to estimate a cohort's rate of dispersal and non-predatory mortality (see Equation (2.85)) but also as a parameter by which a cohort's expected ingestion rate can be back-calculated, if desired. Estimating specific growth rates for BASS, however, obviously depends on the underlying model used to describe the fish's expected growth rate dynamics (i.e., dW/dt). Selecting an appropriate growth model for use by the BASS simulation software, like most model selections, was not a trivial issue since over the past 50 years at least four different models (i.e., von Bertalanffy, Richards, Gompertz, and Parker-Larkin models) have become standard tools for characterizing the growth of fishes. See Ricker (1979) for a detailed discussion of these models and other less commonly used models.

According to the von Bertalanffy model, a fish's growth rate is the simple mass balance of anabolic processes that are directly proportional to the fish's surface area and of catabolic processes that are directly proportional to the fish's body weight. Consequently, the fish's growth dynamics are governed by the following differential equation

$$\frac{dW_w}{dt} = \phi W_w^{2/3} - \rho W_w \quad (3.15)$$

where ϕ is the fish's rate of feeding and assimilation; and ρ is the fish's total metabolic rate. Assuming isometric growth (i.e., $W_w = \lambda L^3$), this model is also equivalent to

$$\frac{dL}{dt} = \frac{\rho}{3} (L_{\max} - L) \quad (3.16)$$

where L is the fish's body length; and $L_{\max} = \phi / (\rho \lambda^{1/3})$ is the fish's "maximum" body length that is obtained by setting Equation (3.15) to zero. For further discussion, see Parker and Larkin (1959) and Paloheimo and Dickie (1965).

The Richards model (Richards 1959) is a generalization of the von Bertalanffy model that relaxes the assumption of isometric growth and strict proportionality between a fish's feeding/assimilatory processes and its absorptive surface areas. In this model, the fish's feeding is simply assumed to be a power function of its body weight. The fish's growth is then described by the differential equation

$$\frac{dW_w}{dt} = \phi_1 W_w^{\phi_2} - \rho W_w \quad (3.17)$$

Although the von Bertalanffy and the Richards models appear to have strong physiological foundations, a critical analysis of the parameters of these models casts doubts on such assertions. One particular point of contention is the assumption that a fish's metabolism (i.e., respiration and excretion) is directly proportional to its body weight. Although this assumption is certainly satisfied or closely approximated for some fish species, most fish species have metabolic demands that are best described as power functions of their body weights. Consequently, from a purely physiologically-based perspective, a better anabolic-catabolic process model for fish growth would be

$$\frac{dW_w}{dt} = \phi_1 W_w^{\phi_2} - \rho_1 W_w^{\rho_2} \quad (3.18)$$

See Paloheimo and Dickie (1965). Unlike the von Bertalanffy and Richards models, however, this model generally does not have a closed analytical solution. Furthermore, when this model is fit to observed data, there is no a priori guarantee that the fitted exponents will actually match expected physiological exponents unless the analysis is suitably constrained.

In light of these criticisms, simpler empirical growth models may be more than adequate for most applications. Two such models that have proved useful in this regard are the Gompertz and Parker-Larkin models. Both of these models are intended to describe the growth of fishes that decreases with the age or size of the individual. Whereas the Gompertz model describes fish growth by

$$\frac{dW_w}{dt} = \epsilon_1 \exp(-\epsilon_2 t) W_w \quad (3.19)$$

the Parker-Larkin model (Parker and Larkin 1959) simply assumes that

$$\frac{dW_w}{dt} = \alpha W_w^\beta \quad (3.20)$$

where the exponent β is less than 1.

Although each of the aforementioned models can describe very different growth trajectories, much of the discussion surrounding their use has focused on whether the models predict asymptotically zero or indeterminate growth (Parker and Larkin 1959, Paloheimo and Dickie 1965, Knight 1968, Schnute 1981). Although growth rates of individual fish almost always decrease with increasing age or body size, Knight (1968) argued that the traditional notion of asymptotically zero growth is seldom, if ever, supported by studies that have focused on actual growth increments rather than on size-at-age. Because the Parker-Larkin model is the only model outlined above that assumes fish growth is fundamentally

indeterminate, and because the Parker-Larkin model does not rely on the a priori assumption that fish respiration is a linear function of their body weight as does the von Bertalanffy and Richards models, this model is used exclusively by BASS when needed.

Three basic types of data have been traditionally used to calculate fish growth rates; these are: 1) length at age or capture, 2) back-calculated length at age for specific age classes sampled over multiple years, and 3) back-calculated length at age for specific year classes or cohorts. Back-calculated body lengths for the latter two data types are generally calculated by regression using measured growth increments of body scales, otoliths, pectoral spines, or other "hard" structures. Whereas for a length at age dataset each individual fish contributes only one observation (i.e., its current length), each individual fish contributes a time series of body lengths for both of the remaining types of growth data.

To estimate specific growth rates for fish, body lengths at age that have been reported in the literature, whether back-calculated or not, are converted into live body weights using weight-length regressions that were reported by either the study of interest or other published sources. Estimated live body weights are then fit to the analytical solution Parker-Larkin growth model,

$$\frac{dW_w}{dt} = \gamma W_w = \left(g_1 W_w^{-g_2} \right) W_w \quad (3.21)$$

using the NL2SOLV nonlinear optimization software. The explicit solution of the Parker-Larkin growth model for any time interval $[t_0, t]$ is

$$W_w(t) = \left[g_1 g_2 (t - t_0) + W_w(t_0)^{g_2} \right]^{1/g_2} \quad (3.22)$$

However, because this expression is discontinuous at $g_2 = 0$, the growth parameters g_1 and g_2 are actually obtained by fitting calculated body weights to the equivalent expression

$$W_w(t) = \left[g_1 \exp(b) (t - t_0) + W_w(t_0)^{\exp(b)} \right]^{1/\exp(b)} \quad (3.23)$$

where $g_2 = \exp(b)$.

Estimation of Hyperbolic Arrhenius Functions

When a fish's daily rate of maximum food ingestion, plankton filtration, gastric evacuation, respiration, or growth exhibits a temperature optimum, the BASS database generator fits the process's actual or synthetic data to the hyperbolic Arrhenius function

$$P = p_1 W_w^{p_2} \exp(p_3 T) \left(1 - \frac{T}{T_2} \right)^{p_3 (T_2 - T_1)} \quad (3.24)$$

The BASS database generator also fits actual or synthetic data regarding satiation meal size and feeding times to satiation to the above equation when these feeding parameters exhibit temperature optima. Testing of the initial NL2SOL-based procedure developed to estimate the parameters of Equation (3.24) revealed that the convergence performance of NL2SOL could be greatly improved by reconfiguring Equation (3.24) as

$$P = p_1 W_w^{p_2} \exp(p_3 T) \left(1 - \frac{T}{T_{\max} + \delta^2} \right)^{p_3 (T_{\max} + \delta^2 - T_1)} \quad (3.25)$$

where T_{\max} is the maximum temperature of the dataset being fitted, and, therefore, $T_2 = T_{\max} + \delta^2$. Because estimation of nonlinear parameters are frequently sensitive to their required initial estimates, a three-step procedure was developed to estimate the parameters for Equation (3.25).

The first step in this procedure estimates a mean body weight exponent \bar{p}_2 by using repeated linear least-squares regressions

$$\log P_i = p_{2,k} \log W_{w,i} + p_{0,k} \quad (3.26)$$

on data subsets whose maximum range of temperatures is less than 3 Celsius.

The second step of the procedure then uses NL2SOL to estimate the parameters of the reduced model

$$\hat{P} = \frac{P}{W_w^{\bar{p}_2}} = p_1 \exp(p_3 T) \left(1 - \frac{T}{T_{\max} + \delta^2} \right)^{p_3 (T_{\max} + \delta^2 - T_1)} \quad (3.27)$$

To estimate these parameters, multiple sets of initial parameters are sequentially supplied to NL2SOL, and the parameter set that produces the smallest sum of least-squares is used in the third and final step in the estimation process.

Initial parameter estimates for Equation (3.27) are generated by first fitting the cubic polynomial

$$\hat{P} = \hat{p}_3 T^3 + \hat{p}_2 T^2 + \hat{p}_1 T + \hat{p}_0 \quad (3.28)$$

using ordinary linear least-squares techniques. The temperature T_e corresponding to the local maximum of the above cubic polynomial, i.e.,

$$\left. \frac{d\hat{P}}{dT} \right|_{T_e} = 3\hat{p}_3 T_e^2 + 2\hat{p}_2 T_e + \hat{p}_1 = 0 \quad (3.29)$$

$$\left. \frac{d^2\hat{P}}{dT^2} \right|_{T_e} = 6\hat{p}_3 T_e + 2\hat{p}_2 < 0 \quad (3.30)$$

is then assigned as the process's optimum temperature T_1 for

each set of initial parameter values. Initial estimates for δ are then assigned assuming that the fish's upper "tolerance" temperature corresponds to equidistant temperatures within the interval

$$T_{\max} < T_2 < 43 \quad (3.31)$$

Initial estimates of the process's temperature coefficients p_3 are similarly assigned as equidistant values within the interval

$$0.05 < p_3 < 0.75 \quad (3.32)$$

Having assigned initial estimates for p_3 , T_1 , and δ , the process's rate at $T_0 = 0$ is finally assigned as the back-calculated mean

$$p_1 = \frac{1}{n} \sum_{i=1}^n \frac{\hat{P}_i}{\exp(p_3 t_i) (1 - t_i/T_2)^{p_3(T_2 - T_1)}} \quad (3.33)$$

In the third and final step, the results of steps 1 and 2 are supplied to NL2SOL as the "best" initial estimates of the parameters for Equation (3.25). From this step, the final parameters for Equation (3.24) are determined.

Table 3.1 summarizes the results obtained using the aforementioned procedure to estimate maximum daily consumption and maximum meal size for a variety of studies reported in the open literature. **Table 3.2** summarizes the results of converting the maximum daily consumption functions used by the Wisconsin Bioenergetics Model into their "equivalent" hyperbolic Arrhenius form. **Figure 3.1** and **Figure 3.2** display selected results from **Table 3.2**.

Readers interested in obtaining the Fortran 95 subroutines used to implement this procedure can do so by simply requesting this code from the author.

3.5. Suggested Calibration Procedures

Calibrating Fish Growth Rates

Because Equations (3.21) and (3.22) do not explicitly account for either reproductive losses or temperature-dependent growth, growth rates estimated by these equations generally should be calibrated for the application at hand when back-calculating fish ingestion rates from estimated growth rates.

Having estimated a long-term average growth rate

$$\gamma = g_1 W_w^{-g_2} \quad (3.34)$$

for a species of interest, the calibration procedure developed for BASS assumes that the fish's specific growth coefficient g_1

is actually an exponential function the fish's ambient water temperature that, in turn, is assumed to be a sinusoidal function of the time of year. In particular,

$$g_1 = g_0 \exp[g_3 (T_m + \alpha \sin(\beta t + \omega))] \quad (3.35)$$

where $g_3 = 0.1 \ln(Q_{10,G})$ defines the fish's Q_{10} relationship for growth; T_m is the mean annual water temperature experienced by the fish; and α , β , and ω are the coefficients describing the amplitude, frequency, and phase shift of the water temperatures experienced by the fish, respectively. Under this assumption a fish's growth is therefore described by

$$\frac{dW_w}{dt} = \left\{ g_0 W_w^{-g_2} \exp[g_3 (T_m + \alpha \sin(\beta t + \omega))] \right\} W_w \quad (3.36)$$

If t_0 is the day that the species' young-of-year are recruited into the population, and m is the integer age in years when the fish becomes sexually mature, it then follows that a fish's pre-spawn body weight at the time of its first reproduction is given by

$$W_w(t_0 + 365 m)^{g_2} - W_w(t_0)^{g_2} = g_0 g_2 \int_{t_0}^{t_0 + 365 m} \exp[g_3 (T_m + \alpha \sin(\beta \tau + \omega))] d\tau \quad (3.37)$$

However, because the integrand of this equation is a harmonic function possessing an annual period, the preceding equation can be simplified to

$$W_w(t_0 + 365 m)^{g_2} - W_w(t_0)^{g_2} = g_0 g_2 m I \quad (3.38)$$

where

$$I = \int_{t_0}^{t_0 + 365} \exp[g_3 (T_m + \alpha \sin(\beta \tau + \omega))] d\tau \quad (3.39)$$

Once fish reach sexual maturity, their underlying growth equation (i.e., Equation (3.36)) is only piecewise differentiable since fish are assumed to lose a constant fraction (σ) of their body weight during spawning due both to gamete production and increased metabolic expenditures associated with spawning behaviors. If n is the species' maximum integer age in years, Equation (3.36) can be integrated between any two consecutive spawning events $i = m, (m + 1), \dots, (n - 1)$ as follows

$$W_w(t_0 + 365 (i + 1))^{g_2} - [(1 - \sigma) W_w(t_0 + 365 i)]^{g_2} = g_0 g_2 I \quad (3.40)$$

$$W_w(t_0 + 365(i+1))^{g_2} - (1 - \rho) W_w(t_0 + 365i)^{g_2} = g_0 g_2 I \quad (3.41)$$

where $\rho = 1 - (1 - \sigma)^{g_2}$. Summing Equations (3.38) and (3.41) appropriately, it then follows that

$$W_w(t_0 + 365n)^{g_2} + \rho \left(\sum_{i=m}^{n-1} W_w(t_0 + 365i)^{g_2} \right) - W_w(t_0)^{g_2} = g_0 g_2 n I \quad (3.42)$$

To calibrate a species growth rate for a particular application using Equations (3.34), (3.39) and (3.42), one must obviously specify the parameters (T_m , α , β , and ω) describing the application's water temperatures and the species' maximum age ($a_{\max} = 365n$), mean age of sexual maturity ($365m$), annual spawning times ($t = (t_0 + 365m), (t_0 + 365(m+1)), \dots$), reproductive/spawning loss constant (σ), initial body weight of young-of-year fish ($W_w(t_0)$), body weight of fish at maximum age ($W_w(t_0 + 365n)$), and the species' allometric growth exponent (g_2). The species' pre-spawn body weights for Equation (3.41) can be estimated using Equation (3.22) using the adjusted allometric growth coefficient

$$g_1 = \frac{W_w(t_0 + 365n)^{g_2} - W_w(t_0)^{g_2}}{g_2 a_{\max}} \quad (3.43)$$

To demonstrate this calibration procedure, growth rates estimated for brook trout (*Salvelinus fontinalis*) from literature data will now be calibrated for a "typical" Mid-Atlantic trout stream whose annual temperature regime is assumed to be given by

$$T[\text{Celsius}] = 10.8 + 8.8 \sin(0.0172 * t + 6.04) \quad (3.44)$$

This temperature function assumes that the stream's annual range of water temperatures is 2 to 19.5 Celsius, that April 1 corresponds to $t = 0$, and that January 15 is the coldest day of the year. In this stream, brook trout are assumed to be recruited into the population with an initial YOY body weight equal to 0.25 g wet wt/fish and live a maximum of seven years. The maximum size attained by these trout is assumed to be 825 g wet wt/fish (i.e., ≈ 440 mm(TL) assuming $W[g] = 0.148 \times 10^{-4} TL[mm]^{2.93}$). Spawning and recruitment are assumed to occur on October 30. Sexual maturity is reached when trout attain a total body length of 157 mm (i.e., between the ages of 2 and 3 years), and the trout's reproductive loss constant is assumed to equal 0.2 g wet wt/g wet wt/spawn.

Finally, the trout's growth Q_{10} is assumed to equal 2 (i.e., $g_3 = 0.069$). Using data compiled by Carlander (1969), the BASS database analysis program estimated the following specific growth rate for brook trout

$$\gamma = 0.0196 W_w^{-0.455} \quad (3.45)$$

Calibrating this growth rate to predict with the trout's assumed maximum and YOY body weights and maximum age using Equation (3.43) then yields

$$\gamma = 0.0178 W_w^{-0.455} \quad (3.46)$$

When this adjusted growth rate is used to project pre-spawn body weights for Equation (3.42) using Equation (3.22), the specific growth rate of brook trout calibrated for reproductive losses and temperature dependencies is

$$\gamma = 0.0107 W_w^{-0.455} \exp[0.069(10.8 + 8.75 \sin(0.0172 t + 6.04))] \quad (3.47)$$

When specific feeding rates (ϕ_{dd} g dry wt/g dry wt/d) are back-calculated monthly using this equation and standard salmonid metabolic relationships (i.e., food assimilation efficiencies, specific dynamic action (SDA) to ingestion ratios, oxygen consumption rates, respiratory quotients (RQ), and ammonia excretion to oxygen consumption quotients(AO)) as outlined by Barber (2003), the following allometric regression can be calculated

$$\phi_{dd} = 0.0251 W_w^{-0.205} \exp(0.064 T) \quad (n = 84; r^2 = 0.98) \quad (3.48)$$

This regression agrees well with results of Sweka and Hartman (2001) who estimated the maximum consumption of brook trout at 12 Celsius to be

$$\phi_{ww} = 0.13 W_w^{-0.20} \quad (3.49)$$

Taken together, the proceeding equations imply that the realized ingestion rate of brook trout at 12 Celsius would be approximately 42% of their maximum ingestion rate. This result agrees well with that reported by Elliott and Hurley (1998).

A Fortran 95 executable program (BASS_CMM_FSH.EXE) is provided with the BASS simulation software to perform the aforementioned growth calibration procedure and back-calculated feeding rate estimation. See Section 5.6.

Estimating Initial Conditions

Although most fish surveys typically report only either total

species densities (fish/ha) or total species biomass (kg wet wt/ha), such data can be easily converted into BASS initial conditions if one assumes that the recruitment strength for each cohort of observed population density has been relatively constant or has been fluctuating around a long term average. To perform this conversion, BASS's assumed self-thinning model Equation (2.84), is first rewritten as

$$\frac{dN}{N} = -b \frac{dW_w}{W_w} \quad (3.50)$$

This equation can then be reintegrated to obtain

$$\ln \frac{N(t)}{N(t_0)} = -b \ln \left(\frac{W_w(t)}{W_w(t_0)} \right) \quad (3.51)$$

$$N(t) = N(t_0) \exp \left[-b \ln \left(\frac{W_w(t)}{W_w(t_0)} \right) \right]$$

A species total population density can then be estimated by applying Equation (3.51) to each of its cohorts, i.e.,

$$N(t) = \sum_i N_i(t) \quad (3.52)$$

$$N(t) = \sum_i N_i(t - a_i) \exp \left\{ -b \ln \left[\frac{W_{w,i}(t)}{W_{w,i}(t - a_i)} \right] \right\}$$

where N_i , $W_{w,i}$, and a_i denote the density, average wet body weight, and age, respectively, of the i -th cohort. Assuming that each cohort is recruited into the species' total population with the same initial body weight ($W_{w,i}(t - a_i) = W_0$) and population density ($N_i(t - a_i) = N_0$), the preceding equation can be simplified to

$$N(t) = N_0 \sum_i \exp \left\{ -b \ln \left[\frac{W_{w,i}(t)}{W_0} \right] \right\} \quad (3.53)$$

If the growth rate trajectories of each cohort have also remained relatively constant, it follows that an expected decomposition of a species total population density into its component cohort densities would be

$$N(t) = N_0 \sum_i \exp \left\{ -b \ln \left[\frac{W_{w,i}(a_i)}{W_0} \right] \right\} \quad (3.54)$$

It also follows that an expected decomposition of a species

total biomass into its component cohort biomasses would be

$$B(t) = \sum_i W_{w,i}(a_i) N_i(t) \quad (3.55)$$

$$= N_0 \sum_i W_{w,i}(a_i) \exp \left\{ -b \ln \left[\frac{W_{w,i}(a_i)}{W_0} \right] \right\}$$

From Equations (3.54) and (3.55) it should be reasonably clear that given a species total population density (N) or total biomass (B) and given a model for the species body growth (i.e., Equations (3.21) and (3.22)), one can straightforwardly calculate the species' apparent long term year-class strength N_0 . Having done so, one cannot only estimate the species' cohort densities but also convert the species' total population density into its expected total biomass and vice versa.

To corroborate the density-to-biomass conversion procedure outlined above, a database of studies that have reported measured fish densities and associated fish biomasses was compiled from the literature (Miles 1978, Quinn 1988, Reed and Rabeni 1989, Ensign et al. 1990, Buynak et al. 1991, Flick and Webster 1992, Bettoli et al. 1993, Waters et al. 1993, Maceina et al. 1995, Mueller 1996, Allen et al. 1998, Radwell 2000, Dettmers et al. 2001, Pierce et al. 2001, Habera et al. 2004). Reported fish densities were converted into estimated biomasses assuming evenly spaced self-thinning exponents b ranging from -0.5 to -1.0 at 0.025 increments. Reduced major axis (RMA) regressions were then calculated for each assumed self-thinning exponent. The self-thinning exponent that minimized the intercurve area between the calculated RMA regression line and the identity relationship $B_{obs} = B_{est}$ was $b = -0.825$. This regression was

$$\ln B_{obs} = 0.827 \ln B_{est} - 0.0528 \quad (n = 512; r^2 = 0.64) \quad (3.56)$$

$$B_{obs} = 0.949 B_{est}^{0.827}$$

Figure 3.3 displays the data for the regression (3.56) and the identity relationship $B_{obs} = B_{est}$.

In addition to calibrating fish growth rates and back-calculating feeding rates, the auxiliary BASS program BASS_CMM_FSH.EXE described in the preceding section estimates initial body weights and cohort densities for users given a target initial total species density or a target initial total species biomass. See Section 5.6.

Figure 3.1 Selected results for fitting Equation (2.58) to maximum consumption rates calculated by the algorithms and parameters used by the Wisconsin Bioenergetics Model. Observed data corresponds to the maximum daily consumption of fish weighing 1, 25, 50, 75, and 100 g wet wt/fish at seven equally spaced temperatures between 0 Celsius and the fish's upper tolerance limit.

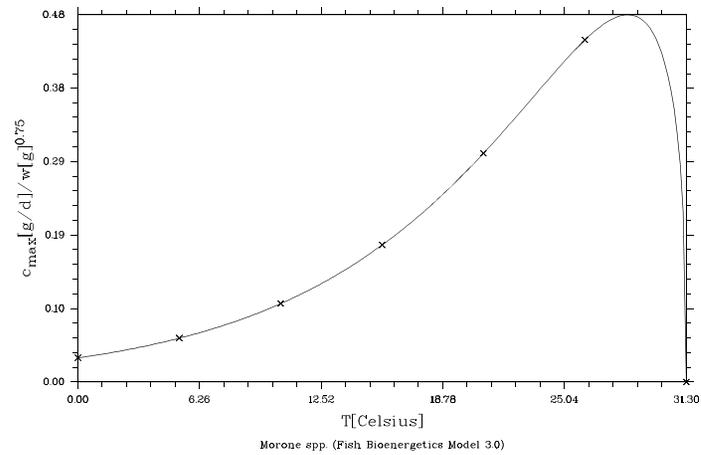
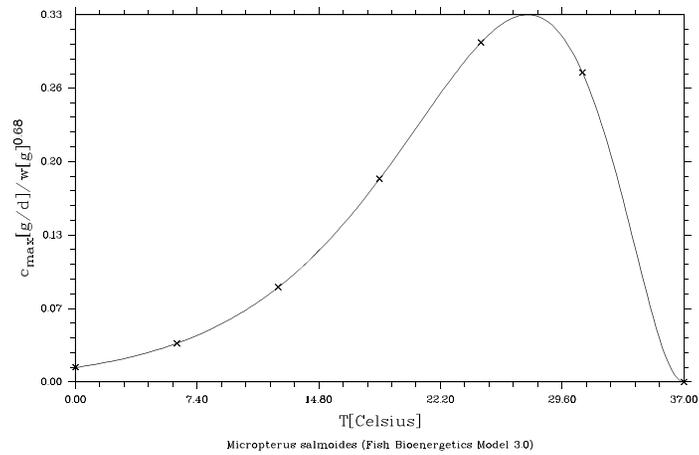
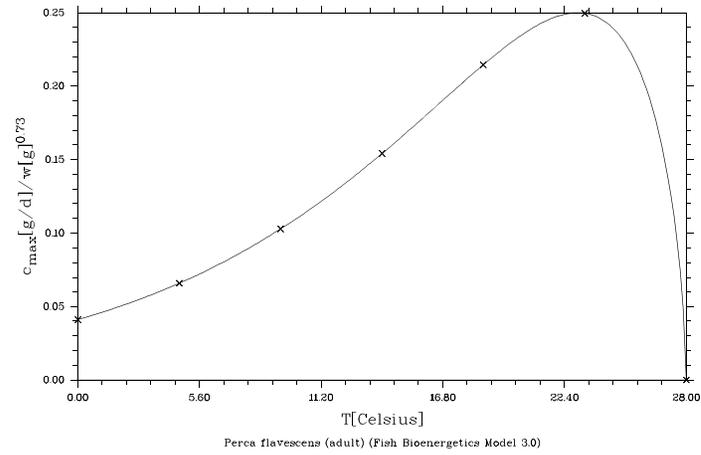
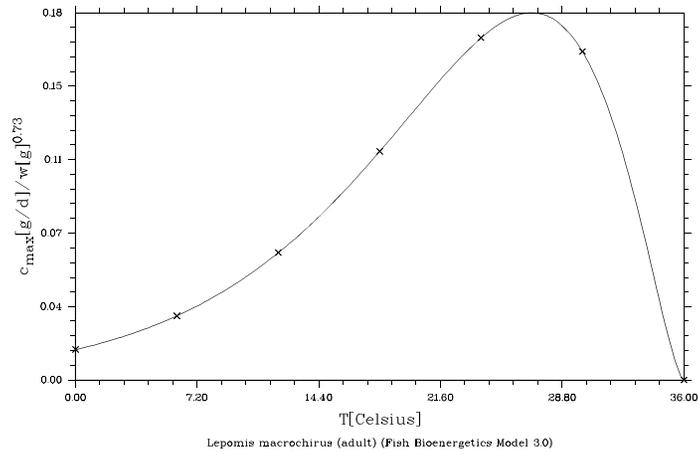


Figure 3.2 Selected results for fitting Equation (2.58) to maximum consumption rates calculated by the algorithms and parameters used by the Wisconsin Bioenergetics Model. Observed data corresponds to the maximum daily consumption of fish weighing 1, 25, 50, 75, and 100 g wet wt/fish at seven equally spaced temperatures between 0 Celsius and the fish's upper tolerance limit.

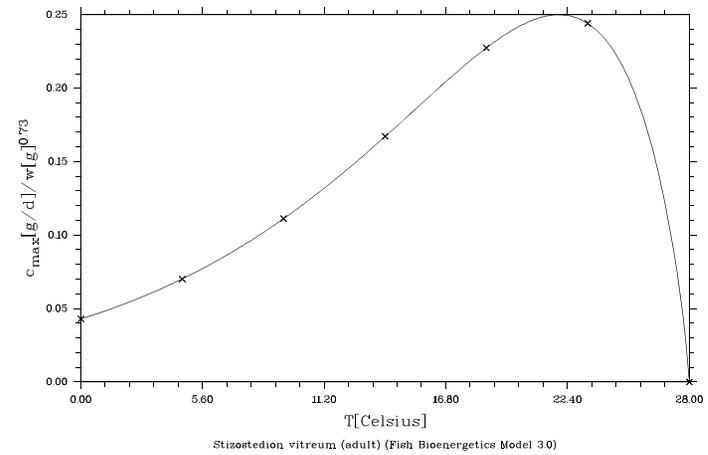
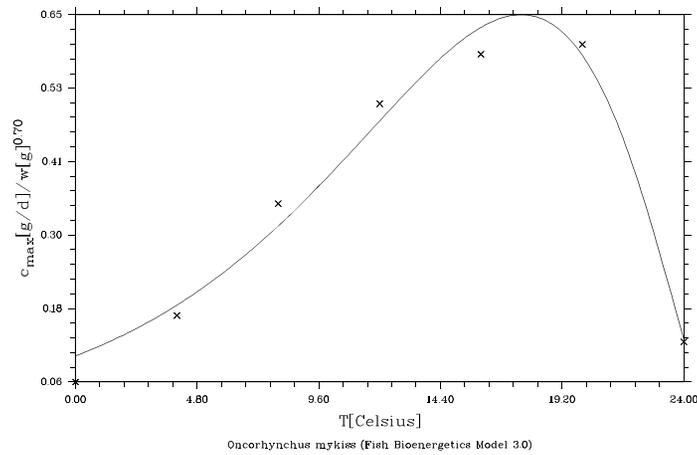
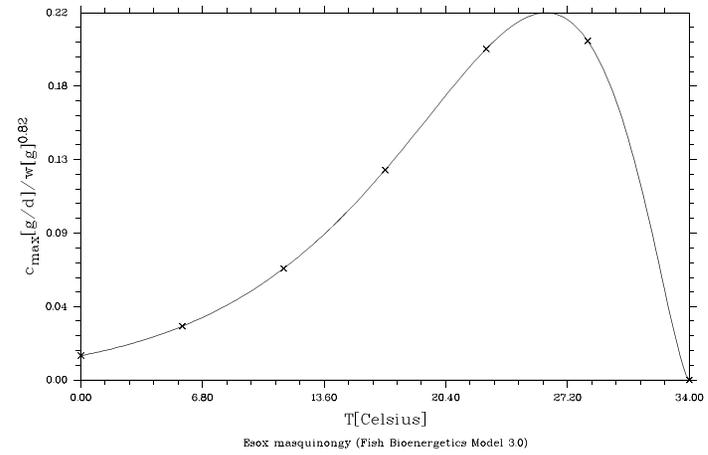
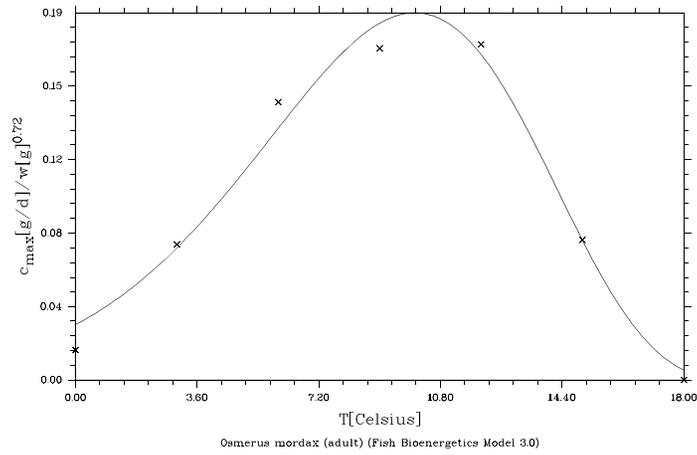


Figure 3.3 Observed fish biomass versus fish biomass predicted by cohort self-thinning BASS's algorithm.

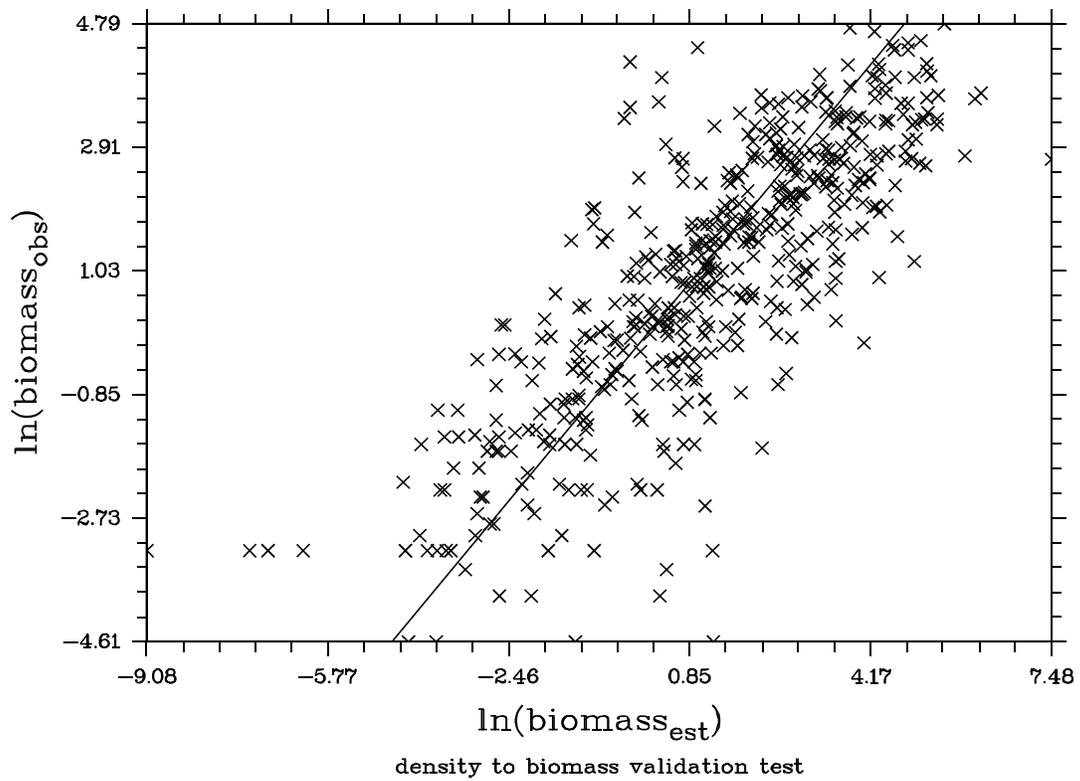


Table 3.1 Summary of NL2SOL regressions for Equation (3.24) fitted to maximum daily consumption rates and satiation meal size reported in the literature.

Species	Process	p_1	p_2	p_3	T_1	T_2	r^2
¹ <i>Channa argus</i>	c_{max} [g/d]	0.00741	0.52	0.425	29.2	51.3	0.99
² <i>Coregonus hoyi</i>	c_{max} [g/g/d]	0.159	-0.54	0.320	16.8	26.0	0.96
³ <i>Morone saxatilis</i>	c_{max} [g/g/d]	0.000945	0.00	0.708	25.9	58.7	0.97
⁴ <i>Morone saxatilis</i>	c_{max} [g/g/d]	0.00542	0.00	0.455	21.6	42.1	0.85
⁵ <i>Pomoxis annularis</i>	c_{max} [g/d]	0.00213	0.03	1.051	23.1	43.0	0.50
⁶ <i>Salmo trutta</i>	c_{max} [Kcal/d]	0.0100	0.76	0.262	18.5	21.8	1.00
⁷ <i>Salmo trutta</i>	sm[mg(dw)]	1.54	0.69	0.596	15.0	29.3	1.00
⁸ <i>Salmo trutta</i>	sm[mg(dw)]	0.731	0.78	2.000	13.8	67.8	1.00
⁹ <i>Salmo trutta</i>	sm[mg(dw)]	0.843	0.76	2.000	13.6	69.5	0.99
¹⁰ <i>Salmo trutta</i>	sm[mg(dw)]	1.72	0.79	0.463	14.9	24.1	1.00
¹¹ <i>Salmo trutta</i>	sm[mg(dw)]	0.906	0.80	0.437	15.1	24.2	0.99
¹² <i>Salvelinus alpinus</i>	c_{max} [g(dw)/g/d]	0.00123	0.00	0.489	16.5	29.0	0.79
¹³ <i>Salvelinus confluentus</i>	c_{max} [g/g/d]	0.00840	0.00	0.288	14.0	29.0	0.98
¹⁴ <i>Siniperca chuatsi</i>	c_{max} [g/d]	0.0267	0.60	0.212	30.3	44.5	0.99
¹⁵ <i>Tilapia zillii</i>	c_{max} [g/g/d]	7.300E-07	0.00	2.000	30.6	75.1	0.94

Data sources and notes

- ¹ Liu et al. (1998). Rates estimated by regression assuming no feeding or lethality at 43 Celsius.
- ² Binkowski and Rudstam (1994). Rates as reported in Table 1 assuming no feeding or lethality at 26 Celsius.
- ³ Cox and Coutant (1981). Rates as reported in Table 2 assuming no feeding or lethality at 43 Celsius.
- ⁴ Hartman and Brandt (1993). Rates estimated from Figure 1 assuming no feeding or lethality at 43 Celsius.
- ⁵ Hayward and Arnold (1996). Rates as reported in Table 1 assuming no feeding or lethality at 43 Celsius.
- ⁶ Elliott (1976b). Rates generated by regressions reported in Table 2.
- ⁷ Elliott (1975). Data as reported in Table 4 for *Baetis*.
- ⁸ Elliott (1975). Data as reported in Table 4 for *Hydropsyche*.
- ⁹ Elliott (1975). Data as reported in Table 4 for chironomids.
- ¹⁰ Elliott (1975). Data as reported in Table 4 for mealworms (*Tenebrio molitor*).
- ¹¹ Elliott (1975). Data as reported in Table 4 for oligochaetes.
- ¹² Larsson and Berglund (1998). Rates as reported in Table 1 assuming no feeding or lethality at 26 Celsius.
- ¹³ Selong et al. (2001). Rates calculated from data reported in Table 2 assuming no assuming or lethality at 26 Celsius.
- ¹⁴ Liu et al. (1998). Rates estimated by regression assuming no feeding or lethality at 43 Celsius.
- ¹⁵ Platt and Hauser (1978). Rates estimated from Figure 1 assuming no feeding or lethality at 43 Celsius.

Table 3.2 Summary of NL2SOL regressions for Equation (2.58) fitted to maximum consumption rates (g wet wt/day) estimated by the Wisconsin Bioenergetics Model 3.0 and its distributed database. Observed data corresponds to the maximum daily consumption of fish weighing 1, 25, 50, 75, and 100 g wet wt/fish at seven equally spaced temperatures between 0 Celsius and the fish's upper tolerance limit.

Species	f_1	f_2	f_3	T_1	T_2	r^2
<i>Alosa pseudoharengus</i> (adult)	0.102	0.70	0.426	15.5	29.3	0.99
<i>Alosa pseudoharengus</i> (juvenile)	0.112	0.70	0.214	19.6	27.3	0.98
<i>Alosa pseudoharengus</i> (yoy)	0.0919	0.70	0.196	21.8	29.2	0.99
<i>Chrosomus</i> spp.	0.0590	0.69	0.094	26.0	29.0	1.00
<i>Clupea harengus</i> (adult)	0.08	0.74	0.644	12.9	29.5	0.99
<i>Clupea harengus</i> (juvenile)	0.0808	0.74	0.535	14.4	31.5	0.99
<i>Coregonus hoyi</i>	0.159	0.46	0.320	16.8	26.0	1.00
<i>Coregonus</i> spp.	0.159	0.68	0.320	16.8	26.0	1.00
<i>Esox masquinongy</i>	0.0147	0.82	0.188	26.0	34.0	1.00
<i>Lates niloticus</i>	0.0112	0.73	0.235	27.5	38.0	1.00
<i>Lepomis macrochirus</i> (adult)	0.0150	0.73	0.172	27.0	36.0	1.00
<i>Lepomis macrochirus</i> (juvenile)	0.0113	0.73	0.138	31.0	37.0	1.00
<i>Micropterus dolomieu</i>	0.00139	0.69	0.296	29.0	36.0	1.00
<i>Micropterus salmoides</i>	0.0129	0.68	0.222	27.5	37.0	1.00
<i>Morone saxatilis</i> (adult)	0.0336	0.75	2.000	21.8	213.9	0.95
<i>Morone saxatilis</i> (age 0)	0.014	0.75	2.000	21.3	153.6	0.99
<i>Morone saxatilis</i> (age 1)	0.0310	0.75	2.000	22.4	221.1	0.98
<i>Morone saxatilis</i> (age 2)	0.0376	0.75	2.000	23.8	268.5	0.96
<i>Morone</i> spp.	0.0314	0.75	0.128	28.3	31.3	1.00
<i>Oncorhynchus gorbuscha</i>	0.142	0.73	0.102	17.0	25.9	0.99
<i>Oncorhynchus kisutch</i>	0.0460	0.73	0.320	15.6	25.8	0.98
<i>Oncorhynchus mykiss</i>	0.102	0.70	0.220	17.6	25.3	0.99
<i>Oncorhynchus nerka</i>	0.142	0.73	0.102	17.0	25.9	0.99
<i>Oncorhynchus tshawytscha</i>	0.0330	0.72	0.230	15.0	18.0	1.00
<i>Osmerus mordax</i> (adult)	0.0304	0.73	0.680	10.0	22.3	0.99
<i>Osmerus mordax</i> (juvenile)	0.0472	0.72	0.207	13.1	18.0	0.98
<i>Osmerus mordax</i> (yoy)	0.0587	0.73	0.143	17.9	26.1	0.98
<i>Perca flavescens</i> (adult)	0.0411	0.73	0.125	23.0	28.0	1.00
<i>Perca flavescens</i> (juvenile)	0.0317	0.73	0.094	29.0	32.0	1.00
<i>Perca flavescens</i> (larvae)	0.0647	0.58	0.094	29.0	32.0	1.00
<i>Petromyzon marinus</i>	0.0766	0.65	0.150	18.0	25.0	1.00
<i>Sarotheradon</i> spp.	0.00643	0.64	0.172	30.0	37.0	1.00
<i>Stizostedion vitreum</i> (adult)	0.0428	0.73	0.138	22.0	28.0	1.00
<i>Stizostedion vitreum</i> (juvenile)	0.0802	0.73	0.094	25.0	28.0	1.00
<i>Theraga chalcogramma</i> (adult)	0.146	0.41	0.270	8	15.0	1.00
<i>Theraga chalcogramma</i> (juvenile)	0.0994	0.41	0.461	8	15.0	1.00

4. BASS User Guide

Although BASS versions 1.0 and 1.1 were written in Fortran 77, BASS version 2.0 and higher are coded in Fortran 95. The model enables users to simulate the population and bioaccumulation dynamics of age-structured fish communities using the temporal and spatial resolution of a day and a hectare, respectively. Although BASS implicitly models the dispersal of fish out of the simulated hectare, it does not explicitly simulate the immigration of fish into the simulated hectare. Monthly or yearly age classes can be used for any species. This flexibility enables users to simulate small, short-lived species such as daces, live bearers, and minnows together with larger, long-lived species such as bass, perch, sunfishes, and trout. The community's food web is specified by defining one or more foraging classes for each fish species based on either body weight, body length, or age. The user then specifies the dietary composition of these foraging classes as a combination of benthos, incidental terrestrial insects, periphyton, phytoplankton, zooplankton, and/or other fish species, including its own. Standing stocks of nonfish compartments can be simulated either as external forcing functions or as state variables.

Although BASS was developed to simulate the bioaccumulation of chemical pollutants within a community or ecosystem context, it can also be used to simulate population and community dynamics of fish assemblages that are not exposed to chemical pollutants. For example, in its present form BASS could be used to simulate the population and community dynamics of fish assemblages that are subjected to altered thermal regimes that might be associated with a variety of hydrological alterations or industrial activities. BASS could also be used to investigate the impacts of exotic species or sport fishery management programs on population or community dynamics of native fish assemblages.

The model's output includes:

- Summaries of all model input parameters and simulation controls.
- Tabulated annual summaries for the bioenergetics of individual fish by species and age class.
- Tabulated annual summaries of chemical bioaccumulation within individual fish by species and age class.
- Tabulated annual summaries for the community level consumption, production, and mortality of each fish species by age class.
- Plotted annual dynamics of selected model variables as

requested by the user.

Please report any comments, criticisms, problems, or suggestions regarding the model software or user manual to

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4.1. General Model Structure and Features

The following features are available in BASS v2.2:

- There are no restrictions to the number of fish species that can be simulated.
- There are no restrictions to the number of cohorts that a fish species can have.
- There are no restrictions to the number of feeding classes that a fish species can have (see the command /FEEDING_OPTIONS).
- There are no restrictions to the number of foraging classes that a fish species can have (see the command /ECOLOGICAL_PARAMETERS).
- There are no restrictions to the number of chemicals that can be simulated.
- Biotransformation of chemicals can be simulated with or without daughter products.
- Integration of BASS's differential equations is performed using a fifth-order Runge-Kutta method with adaptive step sizing that monitors the accuracy of its integration. BASS's Runge-Kutta integrator is patterned on the fifth-order Cash-Karp Runge-Kutta algorithm outlined by Press et al. (1992).

4.2. New Features

The following features were not available in BASS v2.1 and earlier

- Users can now run BASS in a mode that is computationally intermediate between BASS's FGETS and full community modes. In particular, users can simulate fish population dynamics using the conceptual framework of a multi-species Leslie matrix population model. See simulation control command /LESLIE_MATRIX_SIMULATION.
- Benthos, periphyton, phytoplankton, and zooplankton can now be simulated as state variables. To accommodate this capability, the simulation control command /BIOTA in BASS v2.1 has been augmented with five new commands, i.e., /BENTHOS, /TERRESTRIAL_INSECTS, /PERIPHYTON, /PHYTOPLANKTON, and /ZOOPLANKTON. Note, however, BASS v2.2 can still be executed using benthos, periphyton, phytoplankton, and zooplankton as community forcing functions as in BASS v2.1 and earlier.
- Seasonal diets can be specified for any or all foraging classes of a species. See the fish command /ECOLOGICAL_PARAMETERS option **diet(.,.)={...}**.
- The mode of prey body lengths consumed by piscivores can be specified either as a linear or exponential function of their body lengths. See the fish command /ECOLOGICAL_PARAMETERS option **lp[]=fnc**.
- A fish's maximum and minimum prey lengths can now be specified by the user as linear or exponential functions of its body length. See the fish command /ECOLOGICAL_PARAMETERS options **lp_max[]=fnc** and **lp_min[]=fnc**.
- Refuge levels at which cohorts of potential prey species become unavailable to piscivores can now be specified. See the fish command /ECOLOGICAL_PARAMETERS option **refugia[]=fnc**.
- Dispersal and non-predatory mortality are now calculated directly from a fish's expected specific growth rate; the allometric power function formulation used in BASS v2.1 and earlier has been deleted. See the fish command /ECOLOGICAL_PARAMETERS options **nm[]=fnc** and **sg_mu[]=fnc**.
- Size dependent harvest and stocking functions can be specified for any or all species to simulate fisheries management practices. See the fish command /FISHERY_PARAMETERS.
- Habitat suitability indices (HSI) can be specified to adjust a fish's realized feeding/growth, recruitment/spawning, and combined dispersal and non-predatory mortality. See

the fish command /HABITAT_PARAMETERS.

- The syntax for specifying the temperature dependency of a fish's rates of maximum daily consumption, filtering, gastric evacuation, specific growth, and oxygen consumption has been modified. See the fish command /PHYSIOLOGICAL_PARAMETERS.
- An interspecies allometric function is now used to estimate epithelial thicknesses for calculating gill Sherwood numbers and chemical exchange. See Barber (2003)
- Because the new BASS graphical user interface (GUI) enables users to construct their own plots and tables of simulation results, the simulation control commands /ANNUAL_OUTPUTS, /ANNUAL_PLOTS, and /SUMMARY_PLOTS should be considered by most users to be vestigial commands. These commands have been retained for the convenience of model refinement and testing.

4.3. Input File Structure

The general structure of a BASS's input or project file is:

```

/command1 argument(s)
/command2 argument(s)
:
:
/commandn argument(s)
/end

```

The leading slash (/) identifies the line as a command. Blanks or tabs before or after the slash are not significant. The keyword or phrase (i.e., command_n) that follows each slash identifies the type of data being specified by that record. Keywords must be spelled in full without embedded blanks and must be separated from the record's remaining information by at least one blank or tab. Arguments are either integers (e.g., 7), real numbers (e.g., 0, 3.7e-2, 1.3, etc.), or character strings. If the command allows multiple arguments or options, each argument must be separated by a semicolon. Commands can be continued by appending an ampersand (&) to the end of the record; therefore, the following commands are equivalent

```

/command arg1; arg2; arg3; arg4; arg5; arg6

/command arg1; arg2; arg3; &
      arg4; arg5; arg6

```

Because each record is transliterated to lowercase before being decoded, the case of the input file is not significant. Likewise,

because consecutive blanks or tabs are collapsed into a single blank, spacing within a command is not significant. The maximum length of a command line, including continuation lines, is 1024 characters.

An exclamation mark (!) in the first column of a line identifies that line as a comment. An exclamation mark can be also placed elsewhere within a record to start an end-of-line comment, i.e., the remainder of the line, including the exclamation mark, will be ignored.

The last command in any BASS project file must be /END. This command terminates program input and any text or commands following it are ignored. BASS checks the syntactical accuracy of each input command as it is read. If no syntax errors are encountered, BASS then checks the specified input parameters for completeness and internal inconsistency.

BASS input data and commands are broadly classified into four categories: simulation control parameters, chemical parameters, fish parameters, and nonfish biotic parameters. Simulation control parameters provide information that is applicable to the simulation as a whole, e.g., length of the simulation, the ambient water temperature, water column depth, and any desired output options. Chemical parameters specify the chemical's physico-chemical properties (e.g., the chemical's molecular weight, molecular volume, n-octanol / water partition coefficient, etc.) and the chemical's exposure concentrations in various media. Fish parameters specify the fish's feeding and metabolic demands, dietary composition, predator-prey relationships, gill morphometrics, body composition, and initial conditions for the body weights, whole-body chemical concentrations, and population sizes of a fish's cohorts. Nonfish biotic parameters specify how benthos, terrestrial insects, periphyton, and plankton will be simulated.

A BASS project file is actually constructed and managed as a series of include files, i.e., blocks of closely related input commands. These files are specified using the include statement

```
# include 'filename'
```

where *filename* is the name of the file containing the desired commands. Each include file specifies data for either a chemical, a fish species, or a nonfish biotic component. Consequently, a typical BASS project file is structured as follows:

```
! file: bass_input_file.prj
! notes: a BASS project file as specified by include files
!
/command1 simulation control_data
/command2 simulation control_data
```

```
/command3 simulation control_data
# include 'data_for_chemical_1'
# include 'data_for_chemical_2'
# include 'data_for_fish_1'
# include 'data_for_fish_2'
# include 'data_for_fish_3'
# include 'data_for_fish_4'
# include 'data_for_benthos'
# include 'data_for_insects'
# include 'data_for_periphyton'
# include 'data_for_phytoplankton'
# include 'data_for_zooplankton'
/end
```

BASS's graphical user interface (GUI) enables users to create and edit BASS project files and include files in a modular fashion. The actual file structure used by the BASS GUI is detailed in Section 4.5. following the discussion of the BASS input commands themselves below.

4.3.1. Simulation Control Commands

These commands establish the length of the simulation, the ambient water temperature, the community's water level, and other simulation and output options. These data are specified by the following block of commands

/SIMULATION_CONTROL	no argument/option required
/ANNUAL_OUTPUTS	<i>integer</i>
/ANNUAL_PLOTS	<i>string₁; ...; string_n</i>
/BIOTA	<i>string₁; ...; string_n</i>
/FGETS	no argument/option required
/HEADER	<i>string</i>
/LENGTH_OF_SIMULATION	<i>string</i>
/LESLIE_MATRIX_SIMULATION	no argument/option required
/MONTH_T0	<i>string</i>
/NONFISH_QSAR	<i>string₁; ...; string_n</i>
/SUMMARY_PLOTS	<i>string₁; ...; string_n</i>
/TEMPERATURE	<i>string₁; string₂</i>
/WATER_LEVEL	<i>string₁; string₂</i>

Although the command /SIMULATION_CONTROL must be the first command in the block since it identifies the start of these data, the order of the remaining commands is not significant. The use of these commands is described below in alphabetical order.

■ /ANNUAL_OUTPUTS *integer*

This command specifies the time interval, in years, between BASS's annual tabulated and plotted outputs. This number must be a nonnegative integer. BASS assumes a default value of zero that signifies that no annual outputs will be generated. Because

the BASS v2.2 Output Analyzer enables users to generate customized tables and plots, this command is actually a vestigial option of BASS v2.1.

■ /ANNUAL_PLOTS *string*₁; ...; *string*_n

This command specifies the variables whose annual dynamics will be plotted for the years specified by command /ANNUAL_OUTPUTS. Consequently, this command is also a vestigial option of BASS v2.1. Valid options are:

- **afish(variable)** generates plots of each species' total aqueous phase chemical activity as a function of time (day of year) and the species' age, length, or weight class;
- **baf(variable)** generates plots of each species' bioaccumulation factor (i.e., the ratio C_f / C_w) for each chemical as a function of time (day of year) and the species' age, length, or weight class;
- **bmf(variable)** generates plots of each species' biomagnification factor (i.e., the ratio C_f / C_{prev}) for each chemical as a function of time (day of year) and the species' age, length, or weight class;
- **cfish(variable)** generates plots of each species' whole-body concentration (ppm) for each chemical as a function of time (day of year) and the species' age, length, or weight class;
- **pop(variable)** generates plots of each species' population density (fish/ha) as a function of time (day of year) and the species' age, length, or weight class;
- **tl(variable)** generates plots of each species' total body length (cm/fish) as a function of time (day of year) and the species' age or weight class;
- **wt(variable)** generates plots of each species' body weight (g wet wt/fish) as a function of time (day of year) and the species' age or length class.

where *variable* equals "age", "length", or "weight". Each age class or cohort of the species is assigned to one of five size classes that are defined by BASS based on the species' largest / oldest and smallest / youngest individuals.

■ /BIOTA *string*₁; ...; *string*_n

This BASS v2.1 command specifies nonfish standing stocks that are to be generated as forcing functions rather than as simulated state variables. Although this command has been superceded by

the BASS v2.2 commands /BENTHOS, /TERRESTRIAL_INSECTS, /PERIPHYTON, /PHYTOPLANKTON, and /ZOOPLANKTON (see Section 4.3.4), it has been retained for upward compatibility. Valid options are:

- **benthos[yunits] = fnc** to generate benthic standing stocks according to the function *fnc*. The units string *yunits* must be dimensionally equivalent to g dry wt/m².
- **insects[yunits] = fnc** to generate incidental terrestrial insect standing stocks according to the function *fnc*. The units string *yunits* must be dimensionally equivalent to g dry wt/m².
- **periphyton[yunits] = fnc** to generate periphyton standing stocks according to the function *fnc*. The units string *yunits* must be dimensionally equivalent to g dry wt/m².
- **phytoplankton[yunits] = fnc** to generate phytoplankton standing stocks according to the function *fnc*. The units string *yunits* must be dimensionally equivalent to g dry wt/L.
- **zooplankton[yunits] = fnc** to generate zooplankton standing stocks according to the function *fnc*. The units string *yunits* must be dimensionally equivalent to g dry wt/L.

Valid specifications for the function strings *fnc* are :

- **nonfish_name[yunits] = α** to generate a constant compartmental standing stock of α (*yunits*) for the simulation.
- **nonfish_name[yunits] = $\alpha + \beta * \sin(\omega + \phi * t[xunits])$** to generate a sinusoidal compartmental standing stock for the simulation where α is the mean standing stock for the chosen time period, β is its amplitude (*yunits*), ω is its phase angle (radians), and $\phi = 2\pi / \text{period}$ is its frequency ($1/xunits$).
- **nonfish_name[yunits] = file(filename)** to read and interpolate the specified compartmental standing stock from the file *filename*. See Section 4.4.3.

Unless specified otherwise, BASS assumes that the first day of simulation is April 1 and that the 365-th simulation day is March 31. This assignment can be changed using the command /MONTH_T0.

These options are only required when the user is simulating fish that feed on these resources (see the "diet" option for

/ECOLOGICAL_PARAMETERS). Note, however, because BASS assumes that piscivorous fish switch to benthic invertebrates and incidental terrestrial insects when appropriate forage fish are unavailable, the benthos and insect options should be specified even when simulating only piscivorous fish. Also note that if project file uses the FGETS option described below, the only /BIOTA option that might be required is the **zooplankton[yunits]=fnc** option. This option is required only if the user specifies a fish's feeding to be simulated using the clearance model formulation described in Equation (2.64).

If multiple options are selected, each option must be separated by a semicolon.

■ /FGETS

This command enables users to run BASS without simulating the assemblage's population dynamics, i.e., only the growth and bioaccumulation of individual fish are simulated. The command's function and name are based on the FGETS (Food and Gill Exchange of Toxic Substances) model (Barber et al. 1987, 1991) that was BASS's predecessor.

■ /HEADER string

This is an optional command that specifies a title to be printed on each page of the output file. The maximum length of the quoted string is 80 characters.

■ /LENGTH_OF_SIMULATION string

This command specifies the desired length of the simulation. The valid syntax for *string* is

- α [units]

where α is a nonnegative real value. The time unit specified with brackets is converted into days for internal use and subsequent model output.

■ /LESLIE_MATRIX_SIMULATION

This command enables users to run BASS in a mode that is computationally intermediate between BASS's FGETS and full community modes. When this option is specified, BASS simulates fish population dynamics using the conceptual framework of a multispecies Leslie matrix population model. A cohort's mortality is predicted using a single, lumped, self-thinning mortality rate (i.e., Equation (2.84)) without attempting to partition its total mortality into predatory and non-predatory mortality and dispersal as outlined in Sections 2.7 and 2.8. Although predatory mortality is not simulated, the dietary

composition of each cohort is nevertheless predicted using the methods described in Section 2.7. While this simulation option is designed partially to lessen the need for detailed food web information and the work required to calibrate a full community simulation, it is also designed to simulate more realistically the population dynamics of communities in which the dominant process driving cohort mortality and self-thinning is dispersal rather than predation.

■ /NONFISH_QSAR string₁; ...; string_n

This command specifies the quantitative structural activity relationships for the bioconcentration / bioaccumulation factors of the nonfish compartments benthos, periphyton, phytoplankton, and zooplankton that are to be applied to all chemicals. Valid string options are:

- $BCF[-](nonfish_name) = \alpha * Kow[-]^{\beta}$

where $Kow[-]$ is the chemical's n-octanol / water partition coefficient; and α and β are real or integer empirical constants. Also see the chemical command /NONFISH_BCF. When this command is used, the specified QSARs supercede any BCFs specified by /NONFISH_BCF or exposures specified by /EXPOSURE.

■ /MONTH_T0 string

This is an optional command that specifies the month that corresponds to the start of the simulation. If not specified, BASS assumes a default start time of April 1.

■ /SIMULATION_CONTROL

This command specifies the beginning of input data that will apply to the simulation at large, i.e., the length of the simulation and its integration step, the ambient water temperature, community's water level, and various output options.

■ /SUMMARY_PLOTS string₁; ...; string_n

This command specifies the variables whose temporal dynamics will be plotted at the completion of the simulation. This command, like /ANNUAL_PLOTS, is a vestigial option of BASS v2.1. The options can be specified one per card, or all in one card, separated by semicolons. Valid options are:

- **afish(variable)** generates plots of each species' total aqueous phase chemical activity as a function of time (day of simulation) and the species' age, length, or weight class;
- **baf(variable)** generates plots of each species'

bioaccumulation factor (i.e., the ratio C_f / C_w) for each chemical as a function of time (day of simulation) and the species' age, length, or weight class;

- **bmf(variable)** generates plots of each species' biomagnification factor (i.e., the ratio C_f / C_{prey}) for each chemical as a function of time (day of simulation) and the species' age, length, or weight class;
- **cfish(variable)** generates plots of each species' whole-body concentration (ppm) for each chemical as a function of time (day of simulation) and the species' age, length, or weight class;
- **pop(variable)** generates plots of each species' population density (fish/ha) as a function of time (day of simulation) and the species' age, length, or weight class;
- **tl(variable)** generates plots of each species' total body length (cm/fish) as a function of time (day of year) and the species' age or weight class;
- **wt(variable)** generates plots of each species' body weight (g wet wt/fish) as a function of time (day of year) and the species' age or length class.

where *variable* equals "age", "length", or "weight". Each cohort of the species is assigned to one of five size classes that are defined by BASS based on the species' largest / oldest and smallest / youngest individuals.

■ /TEMPERATURE *string*₁; *string*₂

This command specifies a community's ambient water temperatures. For an unstratified water body only one string option is specified. In this case valid options for this command are:

- **temp[celsius]= α** generates a constant ambient water temperature for the simulation.
- **temp[celsius]= $\alpha + \beta \cdot \sin(\omega + \phi \cdot t[xunits])$** generates a sinusoidal ambient water temperature for the simulation where α is the mean temperature for the chosen time period, β is its amplitude (*yunits*), ω is its phase angle (radians), and $\phi = 2\pi / \text{period}$ is its frequency ($1/xunits$).
- **temp[celsius]=file(filename)** to read and interpolate the ambient water temperature from the file *filename*. See Section 4.4.3.

For a stratified water body, users must specify the temperature of

both the epilimnion and the hypolimnion. In this case valid options are:

- **temp_epilimnion[meter]= α**
- **temp_epilimnion[meter]= $\alpha + \beta \cdot \sin(\omega + \phi \cdot t[xunits])$**
- **temp_epilimnion[meter]=file(filename)**
- **temp_hypolimnion[meter]= α**
- **temp_hypolimnion[meter]= $\alpha + \beta \cdot \sin(\omega + \phi \cdot t[xunits])$**
- **temp_hypolimnion[meter]=file(filename)**

Note that unless specified otherwise BASS assumes that its first day of simulation is April 1 and that the 365-th simulation day is March 31. This assignment can be changed using the command /MONTH_T0.

■ /WATER_LEVEL *string*₁; *string*₂

This command specifies a community's actual water level. For an unstratified water body only one string option is specified. In this case, valid options for this command are:

- **depth[meter]= α** generates a constant water level for the simulation.
- **depth[meter]= $\alpha + \beta \cdot \sin(\omega + \phi \cdot t[xunits])$** generates a sinusoidal water level for the simulation where α is the mean water level for the chosen time period, β is its amplitude (*yunits*), ω is its phase angle (radians), and $\phi = 2\pi / \text{period}$ is its frequency ($1/xunits$).
- **depth[meter]=file(filename)** to read and interpolate the water levels from the file *filename*. See Section 4.4.3.

For a stratified water body, users must specify the depth of both the epilimnion and the hypolimnion. In this case, valid options are:

- **depth_epilimnion[meter]= α**
- **depth_epilimnion[meter]= $\alpha + \beta \cdot \sin(\omega + \phi \cdot t[xunits])$**
- **depth_epilimnion[meter]=file(filename)**
- **depth_hypolimnion[meter]= α**
- **depth_hypolimnion[meter]= $\alpha + \beta \cdot \sin(\omega + \phi \cdot t[xunits])$**
- **depth_hypolimnion[meter]=file(filename)**

Note that unless specified otherwise, BASS assumes that its first day of simulation is April 1 and that the 365-th simulation day is March 31. This assignment can be changed using the command /MONTH_T0.

4.3.2. Chemical Input Commands

The physico-chemical properties and exposure concentrations of each chemical of interest are specified by a block of twelve commands, i.e.,

```

/CHEMICAL string
/EXPOSURE string1; ...; stringn
/LETHALITY string1; ...; stringn
/LOG_AC real number
/LOG_KB1 real number
/LOG_KB2 real number
/LOG_P real number
/MELTING_POINT real number
/METABOLISM string1; ...; stringn
/MOLAR_VOLUME real number
/MOLAR_WEIGHT real number
/NONFISH_BCF string1; ...; stringn

```

The command /CHEMICAL must be the first command in the block since it identifies the start of a new set of chemical parameters. The order of the remaining commands, however, is not significant. The use of these commands will now be described in alphabetical order.

■ /CHEMICAL *string*

This command specifies the start of the input for a new chemical. Each chemical name must be a single character string without embedded blanks or hyphens. If a two-part name is desired, the user should use an underscore “_” as a separating character. This command must precede the commands /EXPOSURE, /LETHALITY, /LOG_AC, /LOG_KB1, /LOG_KB2, /LOG_P, /METABOLISM, /MOLAR_WEIGHT, /MOLAR_VOLUME, and /MELTING_POINT. The name specified by this command is used in conjunction with the command /INITIAL_CONDITIONS to input initial whole-body concentrations of chemicals in each age class of the fish of concern and with the command /METABOLISM to specify daughter products of chemical biotransformation. If the user specifies chemical exposures via the file option, the indicated name is also used to direct reading of the specified exposure files. Otherwise, this name is used only for output purposes; BASS does not use this name to link to any chemical data base.

■ /EXPOSURE *string*₁; ...; *string*_{*n*}

This command enables the user to specify the temporal dynamics of chemical exposures to fish via water or contaminated sediments or via the ingestion of benthic invertebrates, incidental terrestrial insects, or plankton. Exposure concentrations specified by these options are assumed to be completely bioavailable to the fish. For example, water concentrations are assumed to be actual dissolved concentrations and not total water concentrations that include particle-bound chemical. If multiple options are selected,

each option must be separated by a semicolon. Valid options are:

- **cbenthos**[*yunits*]=*func* generates potential dietary exposures to fish via benthic organisms according to the function *func*. Note in BASS 2.1 the six lettered name **cbnth**s was used to specify this exposure function.
- **cinsects**[*yunits*]=*func* generates potential dietary exposures to fish via incidental terrestrial insects according to the function *func*. Note in BASS 2.1 the six lettered name **cinsct** was used to specify this exposure function.
- **cperiphyton**[*yunits*]=*func* generates potential dietary exposures to fish via periphyton according to the function *func*. Note in BASS 2.1 the six lettered name **cphytn** was used to specify this exposure function.
- **cphytoplankton**[*yunits*]=*func* generates potential dietary exposures to fish via phytoplankton according to the function *func*. Note in BASS 2.1 the six lettered name **cpplnk** was used to specify this exposure function.
- **csediment**[*yunits*]=*func* generates sediment exposure concentrations according to the function *func*. Note in BASS 2.1 the six lettered name **csdmnt** was used to specify this exposure function.
- **cwater**[*yunits*]=*func* generates aqueous exposure concentrations according to the function *func*.
- **czooplankton**[*yunits*]=*func* generates potential dietary exposures to fish via zooplankton according to the function *func*. Note in BASS 2.1 the six lettered name **czplnk** was used to specify this exposure function.

The concentration units for each exposure function are specified within the indicated brackets. As previously noted for the simulation control functions, unless specified otherwise, BASS assumes that the first day of simulation is April 1 and that the 365-th simulation day is March 31 for all the time dependent exposure functions discussed in the following. This assignment can be changed using the command /MONTH_T0.

Valid expressions for dietary exposures via benthos, periphyton, phytoplankton, or zooplankton and for benthic sediments are:

- **nonfish_name**[*yunits*]=*a* generates a constant concentration of toxicant in benthos, periphyton, phytoplankton, sediment, or zooplankton.
- **nonfish_name**[*yunits*]=*α****cwater**[*xunits*] generates chemical concentrations in benthos, periphyton,

phytoplankton, sediment, or zooplankton as a chemical equilibrium with the ambient environmental water. If this equilibrium is assumed to be thermodynamic, then the coefficient α generally is equal to the product of the component's dry organic fraction and the chemical's K_{ow} . Also see /NONFISH_BCF.

- **nonfish_name[yunits]=file(filename)** to read and interpolate the concentration of toxicant in benthos, periphyton, phytoplankton, sediment, or zooplankton from the file *filename*. See Section 4.4.3.

Valid expressions for insect dietary exposures are:

- **cinsects[yunits]= α** generates a constant concentration of the toxicant in incidental terrestrial insects.
- **cinsects[yunits]=file(filename)** to read and interpolate the concentration of the toxicant in incidental terrestrial insects from the file *filename*. See Section 4.4.3.

Valid expressions for direct aqueous exposures are:

- **cwater[yunits]= α** generates a constant aqueous concentration for the chemical of concern.
- **cwater[yunits]= α *csediment[xunits]** generates aqueous exposure concentrations as a chemical equilibrium with the benthic sediments. If this equilibrium is assumed to be thermodynamic, then the coefficient α generally is assumed to equal the product of the sediment's organic fraction and the chemical's K_{oc} .
- **cwater[yunits]= α + β *exp(γ *t[xunits])** generates an exponential dissolved chemical water concentration where α and β have units of *yunits* and γ has units of $1/xunits$. This option can be used to simulate a chemical spill or one time application of a pesticide.
- **cwater[yunits]= α + β *sin(ω + ϕ *t[xunits])** generates a sinusoidal dissolved chemical water concentration where α is the mean dissolved chemical water concentration (*yunits*) (over one period), β is the amplitude (*yunits*), ω is its phase angle (radians), and $\phi=2\pi$ / period is its frequency ($1/xunits$). This option might be used to simulate the mobilization of sediment bound contaminants during spring or fall turnover.
- **cwater[yunits]=file(filename)** to read and interpolate the dissolved aqueous concentration of toxicant from the file *filename*. See Section 4.4.3.

Users should be very cautious and judicious when using more than one of the above options since the user can easily construct an exposure scenario that is inconsistent with theoretical constraints on the fate and distribution of contaminants in aquatic systems.

■ /LETHALITY *string*₁; ...; *string*_n

This optional command specifies species-specific LC₅₀'s for the chemicals of concern either as an actual concentration value or as a QSAR function. Valid string options are:

- **LC50[units](fish_name)= α**
- **LC50[units](fish_name)= α *Kow[-]^ β**

where *fish_name* is the common name of the fish species to be simulated; Kow[-] is the chemical's n-octanol / water partition coefficient; and α and β are real or integer empirical constants. BASS converts user supplied LC₅₀'s into their corresponding aqueous chemical activities and then uses the geometric mean of these lethal activities to trigger mortality during the simulation.

If the user desires, simulation of mortality associated with the accumulation of a lethal aqueous chemical activity can be turned-off by using the command line option "-l" as discussed in Section 4.5. When this is done, however, BASS still calculates the fish's total aqueous phase chemical activity and reports it as a fraction of the fish's estimated lethal chemical activity to provide the user with a simple, but useful, monitor of the total chemical status of the fish.

■ /LOG_AC *real number*

This command specifies the log₁₀ of the chemical's aqueous activity coefficient. For organic chemicals, if this parameter is not specified, BASS will estimate the chemical's activity coefficient using its melting point and n-octanol / water partition coefficient.

■ /LOG_KB1 *real number*

This command specifies the log₁₀ of a metal's binding constant for non-lipid organic matter (see Equation (2.6)). This parameter is input only for metals and organometals.

■ /LOG_KB2 *real number*

This command specifies the log₁₀ of a metal's binding constant for refractory organic matter. This parameter is used to calculate metal binding to the fish's dry fecal matter and input only for metals and organometals.

■ /LOG_P *real number*

This command specifies the chemical's $\log_{10} K_{ow}$, where K_{ow} is the n-octanol / water partition coefficient. /LOG_P must be specified for all organic chemicals.

■ /MELTING_POINT *real number*

This command specifies the chemical's melting point (Celsius). This datum, together with the chemical's logP, is used to calculate the aqueous activity coefficient for organic chemicals when that parameter is not specified by the user. See Yalkowsky et al. (1983)

■ /METABOLISM *string₁; ...; string_n*

This optional command specifies species-specific rates of biotransformation for the chemical of concern either as an actual rates or as a QSAR function. Valid string options are:

- BT[units](*fish_name, chemical_name*)= α
- BT[units](*fish_name, chemical_name*)= $\alpha * Kow[-]^{\beta}$
- BT[units](*fish_name, none*)= α
- BT[units](*fish_name, none*)= $\alpha * Kow[-]^{\beta}$

where BT specifies the whole-body-referenced biotransformation rate k_{bt} in Equation (2.47); *fish_name* is the common name of the fish species that can metabolize the chemical of concern; *chemical_name* is the name of the daughter product generated by the metabolism of the chemical of concern; Kow[-] is the chemical's n-octanol / water partition coefficient; and α and β are real or integer empirical constants. If the user does not wish to simulate daughter products because they are insignificant or assumed to be harmless, *chemical_name* can be assigned the value *none*. When daughter products are specified, the user must specify all physico-chemical properties of the identified by-product in the same way that the physico-chemical properties of the parent compound are specified.

■ /MOLAR_VOLUME *real number*

This command specifies the chemical's molecular volume (cm³/mol) that is used to calculate the chemical's aqueous diffusivity, i.e.,

$$D = 2.101 \times 10^{-7} \eta^{1.4} v^{-0.589} \quad (4.1)$$

where D is the toxicant's aqueous diffusivity (cm²/sec); η is the viscosity of water (poise); and v is the chemical's molecular volume (cm³/mol) (Hayduk and Laudie 1974). The viscosity of

water over its entire liquid range is represented with less than 1% error by

$$\log_{10} \frac{\eta_{20}}{\eta_T} = \frac{1.37(T - 20) + 8.36 \times 10^{-4}(T - 20)^2}{109 + T} \quad (4.2)$$

where η_T is the viscosity (centipoise) at temperature T (Celsius), and η_{20} is the viscosity of water at 20 Celsius (1.002 centipoise) (Atkins 1978).

■ /MOLAR_WEIGHT *real number*

This command specifies the chemical's molecular weight (g/mol).

■ /NONFISH_BCF *string₁; ...; string_n*

This command specifies the bioconcentration / bioaccumulation factors for the nonfish compartments benthos, periphyton, phytoplankton, and zooplankton either as a numerical constant or as a QSAR function. Valid string options are:

- BCF[-](*nonfish_name*)= α
- BCF[-](*nonfish_name*)= $\alpha * Kow[-]^{\beta}$

where Kow[-] is the chemical's n-octanol / water partition coefficient; and α and β are real or integer empirical constants. Note that this command or /NONFISH_QSAR must be specified for any nonfish compartment that is simulated as a community state variable.

4.3.3. Fish Input Commands

Model parameters for each fish species of interest are specified by a block of thirteen commands, i.e.,

```
/COMMON_NAME      string
/SPECIES           string
/AGE_CLASS_DURATION string
/SPAWNING_PERIOD  string
/FEEDING_OPTIONS   string1; ...; stringn
/PREY_SWITCHING_OFF
/INITIAL_CONDITIONS string1; ...; stringn
/COMPOSITIONAL_PARAMETERS string1; ...; stringn
/ECOLOGICAL_PARAMETERS string1; ...; stringn
/MORPHOMETRIC_PARAMETERS string1; ...; stringn
/PHYSIOLOGICAL_PARAMETERS string1; ...; stringn
/FISHERY_PARAMETERS string1; ...; stringn
/HABITAT_PARAMETERS string1; ...; stringn
```

The command /COMMON_NAME must be the first command in the block since it is the identifier for the start of a new set of fish

parameters. The order of the remaining commands is not significant. The use of these commands will now be described in alphabetical order.

■ /AGE_CLASS_DURATION *string*

This command is used to specify the duration of each age class. Two character strings, i.e., “month” and “year”, are recognized as valid options.

■ /COMMON_NAME *string*

This command specifies the start of input data for a fish species. The command’s specified common name *string* is used for model output and as a label for specifying the dietary composition of other fish species. Each common name must be a single character string without embedded blanks. If a two-part name is desired, the user should use an underscore “_” as a separating blank. See the **diet** option for the command /ECOLOGICAL_PARAMETERS.

■ /COMPOSITIONAL_PARAMETERS *string*₁; ...; *string*_n

This command specifies aqueous and lipid fractions of the fish. Valid options that must be separated by semicolons are:

- **pa**[-]= $\alpha + \beta * \mathbf{pl}$ [-] specifies the fish’s aqueous fraction as a linear function of the fish’s lipid fraction.
- **pl**[-]= $\alpha * \mathbf{W}[xunits]^\beta$ specifies the fish’s lipid fraction as an allometric function of its body weight. If a fish’s average lipid content is independent of its body weight (i.e., β equals zero), however, this parameter can be specified simply as **pl**[yunits]= α .

where α and β are real or integer empirical constants.

■ /ECOLOGICAL_PARAMETERS *string*₁; ...; *string*_n

This command specifies the ecological parameters that describe the fish’s trophic interactions, non-predatory mortality, and recruitment. Valid options that must be separated by semicolons are:

- **ast_yoy**[-]= $f(\mathbf{b}$ [-]= α , $yoy[xunits]=\beta$, $pop[yunits]=\gamma$) specifies parameters for implementing accelerated self-thinning of young-of-year fish (YOY), or more accurately recently recruited cohorts, that often occurs due to intraspecies competition for territories, refugia, or other habitat resources. The functional argument \mathbf{b} [-]= α specifies the desired accelerated self-thinning exponent. The functional argument $yoy[xunits]=\beta$ defines the age, length, or live weight threshold below which cohorts will

be subject to accelerated self-thinning. Valid expressions for **yoy** are either “**age**”, “**tl**”, or “**wt**”. The final functional argument specifies the population threshold that triggers accelerated self-thinning. Depending on the assumed nature of the competition, this threshold can be specified either as the total density of cohorts satisfying the condition $yoy[xunits] \leq \beta$, or as the total density of cohorts satisfying the condition $yoy[xunits] > \beta$. For the former case, **pop** equals “**pop_yoy**” whereas for the latter case **pop** equals “**pop_adults**”.

- **diet**(*class*, *time*) = { $prey_1 = \epsilon_1, \dots, prey_n = \epsilon_n$ } specifies the dietary composition for fish of the age or size class *class* during the months specified by *time*. The right-hand side of the option specifies the prey items ($prey_n$) and their contribution (ϵ_n) to the fish’s diet. Each $prey_n$ is either the common name of a simulated fish species, “benthos”, “insects”, “periphyton”, “phytoplankton”, or “zooplankton” (see commands /BIOTA and /COMMON_NAME). Depending on its value, ϵ_n is interpreted either as a constant percent contribution or as a prey electivity. In particular, if $1 < \epsilon_n < 100$, then ϵ_n designates the relative frequency of that prey in the fish’s diet independent of its relative abundance in the field. On the other hand, if $-1 < \epsilon_n < 1$, then ϵ_n is considered a prey electivity (see Equation (2.74)). For any given foraging class, a user can specify both constant dietary percentages and prey electivities.

Valid expressions for *class* are:

$\alpha < \mathbf{a}[xunits] < \beta$ for age based dietary classes

$\alpha < \mathbf{l}[xunits] < \beta$ for length based dietary classes

$\alpha < \mathbf{w}[xunits] < \beta$ for weight based dietary classes

where α and β are real or integer empirical constants. Although for a given species all *class* types must be the same (i.e., age, length, or weight), the *class* types between species can be different.

Valid expressions for *time* are either the name of a month or the names of two months separated by a hyphen. For example,

month1, e.g., **july**, or

month1-month2, e.g., **july-december**.

If the diet of a specified age / size class is independent of time of year, “*time*” can be omitted. In this case, *time* =

“january-december” is assumed.

The **diet(.,.)={...}** option can be repeated as many times as needed in order to define a complete lifetime sequence of diets for the fish.

- **lp[yunits]= fnc** specifies the average length of prey consumed by a fish whose body length is $L[xunits]$. Unlike most fish command options, two valid function strings are recognized, i.e.,

$$lp[yunits]=\alpha + \beta*L[xunits] \text{ or}$$

$$lp[yunits]=\alpha + \beta*\exp(\gamma*L[xunits])$$

where α , β , and γ are real or integer empirical constants. If a fish’s average prey size is independent of its body length (i.e., β equals zero), this parameter can be specified simply as **lp[yunits]= α** .

- **lp_max[yunits]= fnc** specifies the maximum length of prey consumed by a fish whose body length is $L[xunits]$. Like the option for a fish’s average prey length, two valid function strings are recognized, i.e.,

$$lp_max[yunits]=\alpha + \beta*L[xunits] \text{ or}$$

$$lp_max[yunits]=\alpha + \beta*\exp(\gamma*L[xunits])$$

where α , β , and γ are real or integer empirical constants. If a fish’s maximum prey size is independent of its body length (i.e., β equals zero), this parameter can be specified simply as **lp_max[yunits]= α** . When this parameter is not specified by the user, BASS assigns the default value **lp_max[cm]=0.5*L[cm]**.

- **lp_min[yunits]= fnc** specifies the minimum length of prey consumed by a fish whose body length is $L[xunits]$. Like the option for a fish’s average prey length, two valid function strings are recognized, i.e.,

$$lp_min[yunits]=\alpha + \beta*L[xunits] \text{ or}$$

$$lp_min[yunits]=\alpha + \beta*\exp(\gamma*L[xunits])$$

where α , β , and γ are real or integer empirical constants. If a fish’s minimum prey size is independent of its body length (i.e., β equals zero), this parameter can be specified simply as **lp_min[yunits]= α** . When this parameter is not specified by the user, BASS assigns the default value **lp_min[cm]=0.1*L[cm]**.

- **mls[yunits]= α** specifies the species’ maximum longevity or life span.

- **nm[-]= $\alpha*b(\beta:\gamma)*sg_mu[-]$** specifies a cohort’s rate of dispersal and non-predatory mortality as a function of its habitat suitability and its long-term specific growth rate **sg_mu[-]**. Whereas α specifies the fraction of the species’ total “mortality” that is attributable to dispersal and non-predatory mortality, β and γ specify the species’ minimum and maximum self-thinning exponents, respectively. See Equations (2.85) and (2.88). If the user elects not to simulate habitat effects on dispersal and non-predatory mortality, this parameter can be specified simply as

$$nm[-] = \alpha*b(\beta)*sg_mu[-]$$

where β is the species’ average self-thinning exponent. Also see the /ECOLOGICAL_PARAMETERS option **sg_mu[]**.

- **rbi[-]= α** specifies the species’ reproductive biomass investment (i.e., grams gametes per gram spawning fish) where α is real empirical constant.

- **refugia[yunits]= α** specifies a refuge population size for each cohort that can be prey for community piscivores where α is real or integer constant. **Yunits** must be dimensionally equivalent to fish/ha. If not specified, BASS assumes no refuge level (i.e., **refugia[yunits]=0**)

- **sg_mu[yunits]= $\alpha*W[xunits]^{\beta}$** specifies the species’ long-term mean specific growth rate where α and β are real or integer empirical constants. **yunits** must be dimensionally equivalent to day^{-1} , and **xunits** must be dimensionally equivalent to g wet wt/fish. If not specified, BASS can estimate this parameter provided that the user specifies the species’ expected body weight at its maximum age. See /ECOLOGICAL_PARAMETERS option **wt_max[]** for details.

- **tl_r0[yunits]= α** specifies the species’ minimum total length when sexual maturity is reached where α is a real or integer empirical constant.

- **wl[yunits]= $\alpha*L[xunits]^{\beta}$** specifies the fish’s live weight as an allometric function of its total length where α and β are real or integer empirical constants.

- **wt_max[yunits]= α** specifies the species’ expected live body weight at its maximum age where α is a real or integer empirical constant. This parameter is required only when the user has not specified the species’ long-term mean specific growth rate using the

/ECOLOGICAL_PARAMETERS option **sg_mu**[]. When **sg_mu**[] is not specified, BASS will estimate the species' long-term specific growth rate based on its maximum life span **mls**[], young-of-year body weight **yoy**[], and **wt_max** []. If the user has specified the species' temperature/seasonal dependent specific growth rate

$$sg[yunits]=\alpha*W[xunits]^{\beta}*H(\gamma,T_1,T_2)$$

(see /PHYSIOLOGICAL_PARAMETERS option **sg** []), BASS estimates the species's long-term specific growth rate by

$$sg_mu[1/d] = \bar{\alpha} * W^{\beta}$$

where $\bar{\alpha}$ is back-calculated as outlined by Equation (3.43). If **sg** [] has not been specified, BASS estimates the species's long-term specific growth rate by

$$sg_mu[1/d] = \bar{\alpha} * W^{(-0.732)}$$

where $\bar{\alpha}$ is back-calculated as outlined by Equation (3.43) using the mean interspecies specific growth exponent (i.e., -0.732) estimated from the BASS model database. Also see Barber (2003).

- **yoy**[*yunits*]= α specifies the live weight of fish recruited into the population as age class 1 where α is a real or integer empirical constant.

■ /FEEDING_OPTIONS *string*₁; ...; *string*_n

This command instructs BASS how to calculate ingestion for a particular age or size range of fish. Valid options for this command are :

- **allometric**(*class*) to model expected feeding using Equation (2.58).
- **clearance**(*class*) to model expected feeding using Equation (2.64).
- **holling**(*class*) to model expected feeding using Equation (2.59).
- **linear**(*class*) to model expected feeding using Equation (2.66).

Valid expressions for *class* are:

$\alpha < a[xunits] < \beta$ if the fish's age determines its feeding algorithm;

$\alpha < l[xunits] < \beta$ if the fish's length determines its feeding algorithm;

$\alpha < w[xunits] < \beta$ if the fish's weight determines its feeding algorithm.

where α and β are real or integer empirical constants. Although for a given species all class types must be the same type (i.e., age, length, or weight), class types between species can be different. The parameters for these models are specified using the /PHYSIOLOGICAL_PARAMETERS command.

■ /FISHERY_PARAMETERS *string*₁; ...; *string*_n

This command specifies stocking and harvest rates for sport fishes. Valid options for this command are:

- **stocking**[*sunits*](**age**[*aunits*]= α , **size**[*bunits*]= β , **season=***time*, **frequency =** *schedule*) = γ specifies the stocking rate of fish of age α and body size β (i.e., total length or live weight) during the time interval specified by *time* and the stocking frequency specified by *schedule*. The units of the specified stocking rate *sunits* must be dimensionally equivalent to fish/ha. Valid expressions for *time* are given below. Valid options for *schedule* are: "weekly", "biweekly", "monthly", or "one_time". If *schedule = weekly*, then a new cohort of γ individuals is added to the species' population once a week throughout the specified period. If *schedule = monthly*, then a new cohort of γ individuals is added to the species' population once a month throughout the specified period.
- **harvest**[*hunits*]($\alpha < L[units] < \beta$, *time*) = γ specifies the fractional harvest rate of fish of the specified length class during the indicated time period. The units of the specified harvest rate *hunits* must be dimensionally equivalent to 1/d.

Valid expressions for *time* are two month-day combinations separated by a hyphen. For example,

march 15 - september 1.

Both of these fishery options can be repeated as many times as needed to define the species stocking and harvest. Additionally, only nonzero stocking and harvesting rates need to be specified.

Fishing mortality and harvest can be turned off without deleting the user's harvest parameters by using the command line option "-f". Similarly, stocking can be turned off without deleting the user's stocking parameters using the command line option "-s".

See Section 4.5 for details.

■ **/HABITAT_PARAMETERS** *string*₁; ...; *string*_n

This command specifies habitat preferences, tolerances, and suitability indices for the species.

Valid options for habitat preferences are:

- **tpref[celsius](class) = γ** specifies the preferred or optimum temperature of the age or size class specified within the parentheses.

Valid expressions for *class* are:

$$\begin{aligned} \alpha < a[xunits] < \beta \\ \alpha < l[xunits] < \beta \\ \alpha < w[xunits] < \beta \end{aligned}$$

where α and β are real or integer empirical constants. This option can be specified repeatedly as needed. Although for a given species all class types must be the same type (i.e., age, length, or weight), class types between species can be different.

Valid options for habitat suitability multipliers are:

- **hsi_feeding[-] = *func*** specifies the species' HSI for feeding by the time function *func*. This HSI is used as a simple linear multiplier on a cohort's maximum ingestion rate when feeding is modeled with either an allometric, Holling, or clearance volume formulation. When a cohort's ingestion is back-calculated from its expected growth rate, the specified HSI is used as a simple linear multiplier on the cohort's specific growth rate. See Section 2.9.
- **hsi_recruitment[-] = *func*** specifies the species's HSI for recruitment by the time function *func*. This HSI is used as a simple linear multiplier on the species' YOY recruitment. See Section 2.9.
- **hsi_survival[-] = *func*** specifies the species' HSI for dispersal and non-predatory mortality by the time function *func*. This HSI is used to control the species' self-thinning exponent that determines, in combination with the fish's growth rate, a cohort's estimated dispersal and non-predatory mortality rate. See Section 2.9.

Valid expressions for these HSI functions are:

- **hsi_name[-] = α** generates a constant HSI for the entire simulation.

- **hsi_name[-] = file(filename)** generates time-varying HSIs either by reading and interpolating HSIs specified by the file *filename* or by reading and interpolating habitat variables and then calculating HSIs using user-supplied logistic regressions. See Sections 4.4.3.

When HSI multipliers are calculated using user-supplied logistic regressions, the desired regressions are specified using the following options:

- **hsi_feeding_equation[-] = regression**
- **hsi_recruitment_equation[-] = regression**
- **hsi_survival_equation[-] = regression**

where *regression* specifies a linear combination of habitat variables X_i that are transformed or raised to an integer or real power. Transformations recognized by BASS include:

$$\begin{aligned} \text{LN}(X_i) &\Rightarrow \ln X_i = \log_e X_i \\ \text{LN}_1(X_i) &\Rightarrow \ln(X_i + 1) = \log_e(X_i + 1) \\ \text{LOG}(X_i) &\Rightarrow \log(X_i) = \log_{10}(X_i) \\ \text{LOG}_1(X_i) &\Rightarrow \log(X_i + 1) = \log_{10}(X_i + 1) \\ \text{SQRT}(X_i) &\Rightarrow \sqrt{X_i} \\ \text{ASIN_SQRT}(X_i) &\Rightarrow \arcsin(\sqrt{X_i}) \\ \text{ASIN_SQRT_PCT}(X_i) &\Rightarrow \arcsin(\sqrt{0.01 X_i}) \end{aligned}$$

Habitat variables must be specified with units inclosed within brackets, and must match in name and units to column variables specified by the data file *filename*. After evaluating the specified logistic regression, BASS calculates the fish's HSI multiplier using the standard equation

$$\text{hsi_name} = 1 / (1 + \text{EXP}(-\text{hsi_name_equation}))$$

When HSI functions are not specified by the user, BASS assigns the default value of 1 to each unspecified HSI function.

■ **/INITIAL_CONDITIONS** *string*₁; ...; *string*_n

This command specifies the species' initial ages, whole-body chemical concentrations, live body weights, and population sizes. Valid options for this command are:

- **age[units] = { x_1, \dots, x_n }** to initialize the age of each cohort with the specified vector. The units that are delineated by brackets must be dimensionally equivalent to days.

- **chemical_name[units]**={ x_1, \dots, x_n } to initialize the whole-body concentration of each cohort for the named chemical by the specified vector. Each name must correspond exactly to a name specified by one of the /CHEMICAL commands. The units of measurement that must be enclosed by brackets must be dimensionally equivalent to $\mu\text{g/g}$ wet wt.
- **wt[units]**={ x_1, \dots, x_n } to initialize the body size of each age class with the specified vector. The units delineated by brackets must be dimensionally equivalent to g wet wt/fish.
- **pop[units]**={ x_1, \dots, x_n } to initialize the population density of each age class with the specified vector. The units delineated by brackets must be dimensionally equivalent to fish/ha.

■ /MORPHOMETRIC_PARAMETERS *string*₁; ...; *string*_n

This command specifies the species' morphometric parameters that are needed to describe the exchange of chemicals across its gills. Each *string* specifies a required morphometric parameter as a simple allometric power function of the fish's body weight. Valid options, which must be separated by semicolons, are:

- **ga[yunits]**= $\alpha * W[xunits]^\beta$ specifies the fish's total gill surface area where α and β are real or integer empirical constants. *yunits* must be dimensionally equivalent to cm^2 or cm^2/g wet wt.
- **id[yunits]**= $\alpha * W[xunits]^\beta$ specifies the interlamellar distance between adjacent lamellae where α and β are real or integer empirical constants. *yunits* must be dimensionally equivalent to cm or cm/g wet wt.
- **ld[yunits]**= $\alpha * W[xunits]^\beta$ specifies the density of secondary lamellae on the primary gill filaments (i.e., number of lamellae per mm gill filament) where α and β are real or integer empirical constants.
- **ll[yunits]**= $\alpha * W[xunits]^\beta$ specifies the fish's lamellar length where α and β are real or integer empirical constants. *yunits* must be dimensionally equivalent to cm or cm/g wet wt.

Note that if the exponent β equals zero for any of these parameters, the resulting term $W[xunits]^0$ does not have to be specified.

■ /PHYSIOLOGICAL_PARAMETERS *string*₁; ...; *string*_n

This command specifies the species' physiological parameters for simulating growth. Each *string* specifies a physiological parameter of the fish as a constant or temperature-dependent power function of its body weight. In particular,

- **ae_plant[-]**= α specifies the fish's assimilation efficiency for periphyton and phytoplankton where α is a real empirical constant less than or equal to one.
- **ae_invert[-]**= α specifies the fish's assimilation efficiency for benthos, insects, and zooplankton where α is a real empirical constant less than or equal to one.
- **ae_fish[-]**= α specifies the fish's assimilation efficiency for fish where α is real a empirical constant less than or equal to one.

- **ge[yunits]**= $\alpha * G[xunits]^\beta * H(\gamma, T_1, T_2)$ specifies the fish's gastric evacuation where G is the mass of food resident in the intestine, and where α , β , γ , T_1 , and T_2 are real or integer empirical constants. *yunits* must be dimensionally equivalent to g dry wt/d. In general, $\gamma=1/2, 2/3, \text{ or } 1$ (Jobling 1981). This parameter is required only if the feeding option **holling**(\cdot) is used.

- **kf_min[-]**= α specifies the minimum condition factor for a fish's continuing existence. In BASS, a fish's condition factor is defined by the ratio

$$kf = \frac{W_w}{\alpha L^\beta} \quad (4.3)$$

where W and L are the fish's current live body weight and total length, respectively; and α and β are the coefficient and exponent for the fish's weight-length relationship (see /PHYSIOLOGICAL_PARAMETERS option **wl**[\cdot]).

- **mf[yunits]**= $\alpha * W[xunits]^\beta * H(\gamma, T_1, T_2)$ specifies the fish's maximum filtering rate where α , β , γ , T_1 , and T_2 are real or integer empirical constants. *yunits* must be dimensionally equivalent to L/d . This parameter is required only if the feeding option **clearance**(\cdot) is used.
- **mi[yunits]**= $\alpha * W[xunits]^\beta * H(\gamma, T_1, T_2)$ specifies the fish's maximum ingestion where α , β , γ , T_1 , and T_2 are real or integer empirical constants. *yunits* must be dimensionally equivalent to g dry wt/d. This parameter is required only if the feeding option **allometric**(\cdot) is used.
- **rq[-]**= α specifies the fish's respiratory quotient; (i.e., $L(\text{CO}_2)$ respired/ $L(\text{O}_2)$ consumed) where α is a real empirical constant.

- **rt:std[-]= α** specifies the ratio of a fish's routine respiration to its standard respiration where α is a real empirical constant. If not specified by the user, BASS assigns a default value equal 2.
- **sda:in[-]= α** specifies the ratio of a fish's SDA to its ingestion where α is a real empirical constant. If not specified by the user, BASS assigns a default value equal 0.17.
- **sg[yunits]= $\alpha * W[xunits]^\beta * H(\gamma, T_1, T_2)$** specifies the fish's specific growth rate where α , β , γ , T_1 , and T_2 are real or integer empirical constants. *yunits* must be dimensionally equivalent to day⁻¹. This parameter is required only if the feeding option **linear(·)** is used.
- **sm[yunits]= $\alpha * W[xunits]^\beta * H(\gamma, T_1, T_2)$** specifies the size of the satiation meal consumed during the interval (0, st) where α , β , γ , T_1 , and T_2 are real or integer empirical constants. *yunits* must be dimensionally equivalent to g dry wt/d. See option **st[yunits]** below. This parameter is required only if the feeding option **holling(·)** is used.
- **so[yunits]= $\alpha * W[xunits]^\beta * H(\gamma, T_1, T_2)$** specifies the fish's standard oxygen consumption where α , β , γ , T_1 , and T_2 are real or integer empirical constants. *yunits* must be dimensionally equivalent to mg O₂/hr or mg O₂/g wet wt/hr.
- **st[yunits]= $\alpha * W[xunits]^\beta * H(\gamma, T_1, T_2)$** specifies the time to satiation when feeding with an initially empty stomach where α , β , γ , T_1 , and T_2 are real or integer empirical constants. See option **sm[yunits]** above. This parameter is required only if the feeding option **holling(·)** is used.

For the options **ge[yunits]**, **mf[yunits]**, **mi[yunits]**, **sg[yunits]**, **sm[yunits]**, **so[yunits]**, and **st[yunits]**,

$$H(\gamma, T_1, T_2) = \exp(\gamma T) \left(1 - \frac{T}{T_2} \right)^{\gamma(T_2 - T_1)} \quad (4.4)$$

where T_1 is the temperature at which each particular process's rate is maximum and T_2 is the upper temperature at which the process is no longer operative. If the process does not exhibit a temperature optimum, then the hyperbolic function **H(γ, T_1, T_2)** should be substituted with the exponential function **exp($\gamma * T$ [celsius])**. Consequently, each of these temperature-dependent power functions can also be specified as

$$\alpha * W[xunits]^\beta * \exp(\gamma * T[\text{celsius}])$$

As noted for the fish's morphometric parameters, if the exponent

β equals zero for any of these temperature-dependent power functions, the term **W[xunits] ^{β} 0** does not have to be specified.

If a required parameter is not specified, the program will terminate with an appropriate error message.

■ /PREY_SWITCHING_OFF

This command disables BASS's prey switching algorithms when a cohort's expected feeding level cannot be satisfied using the dietary compositions specified by the user. By default, BASS's prey switching algorithms are enabled.

■ /SPAWNING_PERIOD *string*

This command specifies the months during which spawning occurs. Valid character strings for this command are either the name of a month or the names of two months separated by a hyphen. For example,

/SPAWNING_PERIOD may

OR

/SPAWNING_PERIOD april-june

The names of the months must be spelled-out in full.

■ /SPECIES *string*

This command specifies the scientific name (genus and species) of the fish to be modeled. When this command is encountered, BASS uses the specified scientific name to assign default ecological, morphological, and physiological parameters for the species of interest. These default parameters are then updated with the data that the user inputs via the /ECOLOGICAL_PARAMETERS, /MORPHOMETRIC_PARAMETERS, and /PHYSIOLOGICAL_PARAMETERS commands. This option, however, is not currently operational in BASS v2.2.

4.3.4. Nonfish Input Commands

These commands specify simulation parameters for benthos, periphyton, incidental terrestrial insects, phytoplankton and zooplankton. The syntax for these commands is as follows

/BENTHOS	<i>string</i> ₁ ; ...; <i>string</i> _{<i>n</i>}
/TERRESTRIAL_INSECTS	<i>string</i> ₁ ; ...; <i>string</i> _{<i>n</i>}
/PERIPHYTON	<i>string</i> ₁ ; ...; <i>string</i> _{<i>n</i>}
/PHYTOPLANKTON	<i>string</i> ₁ ; ...; <i>string</i> _{<i>n</i>}
/ZOOPLANKTON	<i>string</i> ₁ ; ...; <i>string</i> _{<i>n</i>}

Depending on the options selected, BASS generates the standing stocks of these nonfish compartments either as community forcing functions or as community state variables. Although these compartments can be simulated for any desired community, only those components identified as fish prey are required to be specified (see the `diet(.,.)={...}` option for /ECOLOGICAL_PARAMETERS). Note, however, because BASS assumes that piscivorous fish switch to benthic invertebrates and incidental terrestrial insects when appropriate forage fish are unavailable, the benthos and insect options should be specified even when simulating only piscivorous fish.

When benthos, periphyton, incidental terrestrial insects, phytoplankton or zooplankton are treated as community forcing functions, a single option of the form

- **biomass[yunits]=string**

is specified. Valid expressions for this option are:

biomass[yunits]= α for a constant nonfish standing stock

biomass[yunits]= $\alpha + \beta \sin(\omega + \phi * t[xunits])$ for a sinusoidal nonfish standing stock where α is the mean standing stock for the chosen time period, β is its amplitude (yunits), ω is its phase angle (radians), and $\phi = 2\pi / \text{period}$ is its frequency (1/xunits).

biomass[yunits]=file(filename) to read and interpolate a nonfish standing stock from the file *filename*. See Section 4.4.3.

Whereas *yunits* must be dimensionally equivalent to g dry wt/m² for benthos, incidental terrestrial insects, and periphyton, for phytoplankton and zooplankton *yunits* must be dimensionally equivalent to g dry wt/L. As previously noted, BASS assumes that the first day of simulation is April 1 and that the 365-th simulation day is March 31. This assignment can be changed using the command /MONTH_T0. This command-option combination is equivalent to the BASS v2.1 simulation control command /BIOTA

When benthos, periphyton, phytoplankton or zooplankton are treated as community state variables, the following five options must be specified:

- **initial_biomass[yunits]=number**. This option specifies the initial compartmental standing stock of the designated component and is required to simulate the designated nonfish compartment as a BASS state variable. *yunits* must be dimensionally equivalent to g dry wt/m².

- **mean_weight[yunits]=fnc**. This option specifies the average body weight of individuals within the designated nonfish compartment. This parameter is required to simulate the designated nonfish compartment as a BASS state variable. *yunits* must be dimensionally equivalent to g dry wt/ind. Valid expressions for *fnc* are:

mean_weight[yunits]= α generates a constant average individual body weight for the designated prey.

mean_weight[yunits]= $\alpha + \beta \sin(\omega + \phi * t[xunits])$ generates the average individual body weight of the designated prey as a sinusoidal function of time where α is the mean body weight for the chosen time period, β is its amplitude (yunits), ω is its phase angle (radians), and $\phi = 2\pi / \text{period}$ is its frequency (1/xunits).

mean_weight[yunits]=file(filename) to read and interpolate the average individual body weight of the designated prey from the file *filename*. See Section 4.4.3.

Unless specified otherwise, BASS assumes that the first day of simulation is April 1 and that the 365-th simulation day is March 31. This assignment can be changed using the command /MONTH_T0.

- **ingestion[yunits]= $\alpha * W[xunits]^\beta * H(\gamma, T_1, T_2)$** specifies the ingestion rate of individuals within the designated compartment as a function of their average body weight and temperature where α , β , γ , T_1 , and T_2 are real or integer empirical constants. This parameter is required to simulate either benthos or zooplankton as a BASS state variable. *yunits* must be dimensionally equivalent to g dry wt/d, and *xunits* must be dimensionally equivalent to g dry wt/ind.
- **photosynthesis[yunits]= $\alpha * W[xunits]^\beta * H(\gamma, T_1, T_2)$** specifies the photosynthetic rate of individuals within the designated compartment as a function of their average body weight and temperature where α , β , γ , T_1 , and T_2 are real or integer empirical constants. This parameter is required to simulate either periphyton or phytoplankton as a BASS state variable. *yunits* must be dimensionally equivalent to g dry wt/d, and *xunits* must be dimensionally equivalent to g dry wt/ind. Currently, photosynthesis is not treated as a function of nutrients and light availability.
- **respiration[yunits]= $\alpha * W[xunits]^\beta * H(\gamma, T_1, T_2)$** specifies the specific rate of dry organic matter respiration for the designated compartment as a function of average

individual body weight and temperature where α , β , γ , T_1 , and T_2 are real or integer empirical constants. This parameter is required to simulate the designated nonfish compartment as a BASS state variable. *yunits* must be dimensionally equivalent to g dry wt/d, and *xunits* must be dimensionally equivalent to g dry wt/ind.

Although BASS enables users to simulate benthos, periphyton, phytoplankton or zooplankton as community state variables, incidental terrestrial insects are always treated as a community forcing function.

4.4. Input Data Syntax

4.4.1. Units Recognized by BASS

Most BASS commands require the specification of units (or combination of units) as part of an option. This section describes the syntax for units that are recognized by BASS's input algorithms. The conversion of user-supplied units to those actually used by BASS is accomplished by referencing all units to the MKS system (i.e., meter, kilogram, second). **Table 4.1** and **Table 4.2** summarize prefixes and fundamental units, respectively, that are recognized by BASS's unit conversion subroutines. **Table 4.2** also summarizes the dimensionality and the conversion factor to the MKS system standard unit. **Table 4.3** summarizes units that are recognized by BASS's unit conversion subroutines for specifying ecological, morphometric, and physiological units.

Units and their prefixes can be specified in either upper or lower case. When prefixes are used, there must be no embedded blanks between the prefix and the unit name, e.g., "milligrams" is correct, "milli grams" is incorrect. The circumflex (^) is used to denote exponentiation (e.g., cm² is presented as cm^{^2}). The slash (/) is used to denote division. If multiple slashes are used to specify a unit, they are interpreted according to strict algebraic logic. For example, both "mg/liter", and "mg liter⁻¹" are equivalent specifications. Similarly, the weight specific units "mg/g/day" are "mg g⁻¹ day⁻¹" are equivalent.

4.4.2. User Specified Functions

The following syntax rules apply to specifying these options

- Brackets are used only to delineate units. Dimensionless parameters like assimilation efficiency, lipid fraction, and K_{ow} must be specified with null units "[-]".
- The order of addition and multiplication is not significant. Thus, the following specifications are valid and equivalent.

$$\begin{aligned} \text{temp(celsius)} &= \alpha + \beta * \sin(\omega + \phi * t[xunits]) \Leftrightarrow \\ \text{temp[celsius]} &= \beta \sin(\phi * t[xunits] + \omega) + \alpha \end{aligned}$$

$$\begin{aligned} \text{czplnk}[yunits] &= \alpha * \text{cwater}[xunits] \Leftrightarrow \\ \text{czplnk}[yunits] &= \text{cwater}[xunits] * \alpha \end{aligned}$$

- Options that are temperature-dependent or independent power functions can be specified by log₁₀ or ln transforms. For example, the following options are valid

$$\ln(\text{so}[yunits]) = \alpha + \beta * \ln(W[xunits]) + \gamma * T[\text{celsius}]$$

$$\log(\text{so}[yunits]) = \alpha + \beta * \ln(W[xunits]) + \gamma * T[\text{celsius}]$$

- User-specified functions do not have to be in reduced form. For example, temperature-dependent power functions can be specified with a reference temperature other than 0° Celsius. Thus, BASS will correctly decode the following functions

$$\text{so}[yunits] = \alpha * W[xunits]^{\beta} * \exp(\gamma * (T[\text{celsius}] - 20))$$

$$\ln(\text{so}[yunits]) = \alpha + \beta * \ln(W[xunits]) + \gamma * (T[\text{celsius}] - 20)$$

$$\log(\text{so}[yunits]) = \alpha + \beta * \log(W[xunits]) + \gamma * (T[\text{celsius}] - 20)$$

- If the temperature dependency is unknown, temperature-dependent power functions can be input for a specific temperature, γ° Celsius, in which case BASS assumes a default $Q_{10}=2$. If this feature is used, the reference temperature must be enclosed by parentheses and follow the units specification of the independent variable. For example, the following specifications are valid

$$\text{so}[yunits](\gamma) = \alpha * W[xunits]^{\beta}$$

$$\ln(\text{so}[yunits](\gamma)) = \alpha + \beta * \ln(W[xunits])$$

$$\log(\text{so}[yunits](\gamma)) = \alpha + \beta * \log(W[xunits])$$

- If either the slope of a linear function or the exponent of a power function is zero, the function can be input as a constant without specifying the expected independent variable. For example, the following specifications are equivalent

$$\text{lp[cm]} = 4.5 \Leftrightarrow \text{lp[cm]} = 4.5 + 0.0 * L[\text{cm}]$$

$$\text{pl[-]} = 0.05 \Leftrightarrow \text{pl[-]} = 0.05 * W[\text{g(fw)}]^0$$

- Operators (^* / +-) may not be concatenated. For example, the following options have invalid syntax

so[mg(o2)/g/hr]=
 $0.1 * \exp(0.0693 * T[\text{celsius}] * W[\text{g}(fw)]^{-0.2})$

ln(so[mg(o2)/g/hr])=
 $-2.30 + 0.0693 * T[\text{celsius}] - 0.2 * \ln(W[\text{g}(fw)])$

The correct syntax for these options would be

so[mg(o2)/g/hr]=
 $0.1 * \exp(0.0693 * T[\text{celsius}] * W[\text{g}(fw)]^{-0.2})$

ln(so[mg(o2)/g/hr])=
 $-2.30 + 0.0693 * T[\text{celsius}] - 0.2 * \ln(W[\text{g}(fw)])$

4.4.3. User Supplied Parameter Files

If the user specifies a file option for the /EXPOSURE, /TEMPERATURE, /WATER_LEVEL, /BIOTA, /BENTHOS, /TERRESTRIAL_INSECTS, /PERIPHYTON, /PHYTOPLANKTON, /ZOOPLANKTON, or /HABITAT_PARAMETERS commands, the designated files must exist and be supplied by the user. The general format of a BASS exposure file allows a user to specify multiple exposure conditions within a single file. Each file record specifies exposure conditions for a specific time. The general format of a BASS exposure file is as follows

```
!
! file: exposure.dat
!
/001  time[units]  ! see ensuing discussion
/C1 string
:  :
/CM string
/START_DATA
V1,1  V1,2  ...  V1,MV  ! comment
V2,1  V2,2  ...  V2,MV  ! comment
:  :  ...  :
VNR,1  VNR,2  ...  VNR,MV  ! comment
```

The records beginning with a slash (/) followed by an integer CJ identify the type of data (time, exposure concentration, temperature, etc.) contained in CJ-th column of each data record. In this example, NR is the total number of data records in the file, MV is the number of variables per record, and C1...CM are the column positions of M exposure variables that are to be read. Note, however, that MV can be greater than CM and that C1...CM need not be consecutively numbered. To simplify the reading of multiple exposure files, BASS requires that "time" be specified as the first column of any user-supplied exposure file. Valid character strings for specifying the remaining data columns include:

bbenthos[units] to input the standing stock of benthic invertebrates;

binsects[units] to input the standing stock of incidental terrestrial insects;

bperiphyton[units] to input the standing stock of periphyton or grazable algae;

bphytoplankton[units] to input the standing stock of phytoplankton;

bzooplankton[units] to input the standing stock of zooplankton;

cbenthos[units](chemical name) to input the concentration of *chemical name* in benthic invertebrates;

cinsects[units](chemical name) to input the concentration of *chemical name* in incidental terrestrial insects;

cperiphyton[units](chemical name) to input the concentration of *chemical name* in periphyton;

cphytoplankton[units](chemical name) to input the concentration of *chemical name* in phytoplankton;

csediment[units](chemical name) to input the sediment concentration of *chemical name*;

cwater[units](chemical name) to input the unbound, aqueous concentration of *chemical name*;

czooplankton[units](chemical name) to input the whole-body concentration of *chemical name* in zooplankton;

depth[units] to input water depth.

hsi_feeding[-](fish name) to input the feeding/growth HSI for the fish species identified within the parentheses.

hsi_recruitment[-](fish name) to input the recruitment/spawning HSI for the fish species identified within the parentheses.

hsi_survival[-](fish name) to input the dispersal/non-predatory mortality HSI for the fish species identified within the parentheses.

temperature[units] to input ambient water temperature.

wbenthos[units] to input the mean body weight of benthic invertebrates;

winsects[units] to input the mean body weight of incidental terrestrial insects;

wperiphyton[units] to input the mean body weight of periphyton or grazable algae;

wphytoplankton[units] to input the mean body weight of phytoplankton;

wzooplankton[units] to input the mean body weight of zooplankton;

If column names other than those listed above are specified, BASS simply ignores them. Data records can be continued by appending an ampersand (&) to the line, e.g., the following data records are equivalent.

```
Vi,1 Vi,2 ... Vij Vij+1 ... Vi,MV
```

```
Vi,1 Vi,2 ... Vij &  
Vij+1 Vij+2 ... Vi,MV
```

File records must be sequenced such that time is nondecreasing (i.e., $t_i \leq t_{i+1}$, $i = 1, 2, \dots, N-1$). The time increment between consecutive records can be either constant or variable. BASS calculates the exposure conditions between specified time points by simple linear interpolation.

4.5. BASS Include File Structure

As mentioned in Section 4.1, BASS's input processing routines allow a BASS project file to be specified using include files of related parameters. This capability is the cornerstone upon which the BASS GUI has been developed.

In order to select an appropriate project / include file hierarchy for implementation in the BASS GUI, careful consideration was given to the perceived needs of researchers and environmental regulators who would routinely analyze and evaluate similar scenarios that might differ either in terms of the chemical exposures of interest or in terms of the communities of concern. For example, the USEPA Office of Pesticide Programs routinely evaluates different pesticides for registration based on their expected fate and effects in series of canonical aquatic habitats / ecosystems. Similarly, the USEPA Office of Pollution Prevention and Toxics evaluates the pre-manufacturing registration of industrial chemicals in much the same way. These considerations suggested that a practical working BASS project / include file hierarchy should be structured as follows:

- All data specifying the bioenergetic, compositional, and morphological parameters for a specific fish species that

can be considered to be independent of the particular community in which the fish resides, should be contained within a single include file that is assigned the reserved extension FSH.

- All data specifying the structure and function of a particular fish community should be contained within a single include file that is assigned the reserved extension CMM. These files should use the aforementioned *.FSH files as include files intervened with the necessary fish commands that are specific to the community of interest. In general, these community-specific fish commands define each species' 1) the dietary composition, 2) initial ages, body weights, population densities, and chemical residues, 3) habitat multipliers, 4) any desired fishery parameters, and 5) any fish commands contained within the specified *.FSH files that the user wants to have superseded or updated.
- All data specifying the physico-chemical properties for a specific chemical of concern should be contained within a single include file that is assigned the reserved extension PRP.
- All data specifying a chemical exposure scenario should be contained within a single include file that is assigned the reserved extension CHM. These files should use the aforementioned *.PRP files intervened with the necessary chemical commands needed to specify each chemical's 1) aqueous concentration, 2) dietary exposures via benthos, insects, periphyton, phytoplankton, and zooplankton, 3) effects concentrations for specific fish, and 4) any relevant rates of biotransformation by specific fish.
- Lastly, all BASS project files should use the aforementioned *.CMM files and *.CHM files to specify the fish community and the chemical exposures of concern. All such project files will be assigned the reserved extension PRJ.

Based on these considerations, the general structure of a BASS project file is as follows:

```
! file: name.prj  
! notes: general structure of a BASS project file  
!  
/ SIMULATION_CONTROL  
/ HEADER <string>  
/ MONTH_TO <string>  
/ LENGTH_OF_SIMULATION  $\alpha$ [year]  
/ TEMPERATURE temp[celsius] = fnc  
/ WATER_LEVEL depth[meter] = fnc  
! specify chemical exposures (if any)  
#include 'exposures.chm'  
! specify fish community  
#include 'community.cmm'
```

```
/ END
```

The chemical exposure scenario file EXPOSURES.CHM specified in this project file has the following general form

```
! file: exposures.chm
! notes: general structure of a chemical
!       exposure scenario file
!
! specify physico-chemical parameters
#include 'chemical_1.prp'
/ EXPOSURE cwater[ppm] = fnc; &
  cbenthos[ppm] = fnc; &
  cinsects[ppm] = fnc; &
  cperiphyton[ppm] = fnc; &
  cphytoplankton[ppm] = fnc; &
  czooplankton[ppm] = fnc
/ NONFISH_BCF &
  bcf[-](benthos) = fnc; &
  bcf[-](periphyton) = fnc; &
  bcf[-](phytoplankton) = fnc; &
  bcf[-](zooplankton) = fnc
/ LETHALITY &
  lc50[units](fish_1) = fnc; &
  lc50[units](fish_2) = fnc
/ METABOLISM &
  bt[units](fish_1,chem_n) = fnc; &
  bt[units](fish_2,chem_n) = fnc
! repeat above chemical data block as needed
:
! end exposures.chm
```

The general structure of the chemical property file CHEMICAL_1.PRP specified in the above exposure scenario file is

```
! file: chemical_1.prp
! notes: general structure of a chemical
!       property file
!
/ CHEMICAL <string>
/ LOG_AC <real number>
/ LOG_P <real number>
/ LOG_KB1 <real number>
/ LOG_KB2 <real number>
/ MOLAR_WEIGHT <real number>
/ MOLAR_VOLUME <real number>
/ MELTING_POINT <real number>
! end chemical_1.prp
```

The community file COMMUNITY.CMM specified for the above project file has the following general form

```
! file: community.cmm
! notes: general structure of a community file
!
#include 'fish_1.fsh'
/ ECOLOGICAL_PARAMETERS &
  diet( $\alpha <x[units]<\beta$ )={benthos= $\alpha$ ,...,fish_n= $\beta$ ,...}; &
  diet( $\alpha <x[units]<\beta$ )={benthos= $\alpha$ ,...,fish_n= $\beta$ ,...}; &
  diet( $\alpha <x[units]<\beta$ )={benthos= $\alpha$ ,...,fish_n= $\beta$ ,...}; &
  diet( $\alpha <x[units]<\beta$ )={benthos= $\alpha$ ,...,fish_n= $\beta$ ,...}; &
  nm[-]= $\alpha * b(\beta:\gamma) * sg\_mu[-]$ ; &
  ast_yoy[-]=f(bb[-]= $\alpha$ , size= $\beta$ , pop= $\gamma$ )
/ INITIAL_CONDITIONS &
  age[yr]={ $\alpha$ ,..., $\beta$ }; &
  wt[g]={ $\alpha$ ,..., $\beta$ }; &
  pop[fish/ha]={ $\alpha$ ,..., $\beta$ }
```

```
/ FISHERY_PARAMETERS &
  stocking[fish/ha] &
  (age[yr]= $\alpha$ , size[g]= $\beta$ , season= $s1$ , frequency= $s2$ )= $\gamma$ ; &
  harvest[l/yr]( $\alpha <l[cm]<\beta$ , season_string)= $\gamma$ 
/ HABITAT_PARAMETERS &
  hsi_feeding[-]=fnc; &
  hsi_recruitment[-]=fnc; &
  hsi_survival[-]=fnc
! repeat above fish data block as needed
:
! specify nonfish compartments/forcing functions
/ BENTHOS &
  initial_biomass[units]= $\alpha$ ; &
  mean_weight[g(dw)]=fnc; &
  ingestion[g(dw)/day]= $\alpha * w[g(dw)]^{\beta} * h(\gamma, t_1, t_2)$ ; &
  respiration[g(dw)/day]= $\alpha * w[g(dw)]^{\beta} * h(\gamma, t_1, t_2)$ 
/ TERRESTRIAL_INSECTS biomass[units]=fnc
:
! end community.cmm
```

The general structure of the fish parameter file FISH_1.FSH specified in the above community file is

```
! file: fish_1.fsh
! notes: general structure of a fish file
!
/ COMMON_NAME <string>
/ SPECIES <string>
/ AGE_CLASS_DURATION <string>
/ SPAWNING_PERIOD <string>
/ FEEDING_OPTIONS &
  allometric( $\alpha <x[units]<\beta$ ); &
  clearance( $\alpha <x[units]<\beta$ ); &
  holling( $\alpha <x[units]<\beta$ ); &
  linear( $\alpha <x[units]<\beta$ )
/ COMPOSITIONAL_PARAMETERS &
  pa[-]= $\alpha * pl[-] + \beta$ ; &
  pl[-]= $\alpha * w[g]^{\beta}$ 
/ ECOLOGICAL_PARAMETERS &
  yoy[g]= $\alpha$ ; wt_max[g]= $\alpha$ ; mls[yr]= $\alpha$ ; &
  tl_r0[cm]= $\alpha$ ; rbi[-]= $\alpha$ ; &
  lp[cm]= $\alpha * l[cm] + \beta$ ; &
  lp_max[cm]= $\alpha * l[cm] + \beta$ ; &
  lp_min[cm]= $\alpha * l[cm] + \beta$ ; &
  sg_mu[l/d]= $\alpha * w[g]^{\beta}$ ; &
  wl[g]= $\alpha * l[cm]^{\beta}$ 
/ MORPHOMETRIC_PARAMETERS &
  ga[cm^2]= $\alpha * w[g]^{\beta}$ ; &
  id[cm]= $\alpha * w[g]^{\beta}$ ; &
  ld[cm]= $\alpha * w[g]^{\beta}$ ; &
  ll[cm]= $\alpha * w[g]^{\beta}$ 
/ PHYSIOLOGICAL_PARAMETERS &
  ae_fish[-]= $\alpha$ ; ae_invert[-]= $\alpha$ ; ae_plant[-]= $\alpha$ ; &
  rq[-]= $\alpha$ ; rt_std[-]= $\alpha$ ; sda:in[-]= $\alpha$ ; &
  ge[g(dw)/d]= $\alpha * g[g]^{\beta} * h(\gamma, t_1, t_2)$ ; &
  mf[l/d]= $\alpha * w[g]^{\beta} * h(\gamma, t_1, t_2)$ ; &
  mi[g(dw)/d]= $\alpha * w[g]^{\beta} * h(\gamma, t_1, t_2)$ ; &
  sg[l/d]= $\alpha * w[g]^{\beta} * h(\gamma, t_1, t_2)$ ; &
  sm[g(dw)]= $\alpha * w[g]^{\beta} * h(\gamma, t_1, t_2)$ ; &
  so[mg(o2)/h]= $\alpha * w[g]^{\beta} * h(\gamma, t_1, t_2)$ ; &
  st[minute]= $\alpha * w[g]^{\beta} * h(\gamma, t_1, t_2)$ 
! end fish_1.fsh
```

In addition to the file hierarchy outlined above, it was also concluded that the BASS software and GUI should operate within a fixed directory structure that would accommodate both project-specific include files and canonical include files that could be copied and edited as needed for new applications. Consequently,

using the UNIX like command line

```
C:\BASS22> bass_v22 -i input.prj
```

Although the “-i filename” option is the only required command line option, the following additional options are available

- c => print distribution of cpu time in major subroutines
- e => output realized monthly dietary compositions for electively feeding fish
- f => turn off fishing mortality
- h => print this help list and stop (also see -?)
- l => calculate the total aqueous phase chemical activity of fish but turn off associated incipient lethality
- m => enable monthly spawning for species with annual age classes
- mba => output mass balance analysis associated with each requested annual summary
- n => internally calculate rate-based BCFs for nonfish (see

Equation (2.102))

- p => turn on messages associated with feeding and predation
- s => turn off fish stocking
- t => run test of the BASS Runge-Kutta integrator and stop; results outputted to file BASS_INT_TEST.OUT.
- w => read project file and generate associated message file without attempting model execution
- z => output ending vectors for age, weight, density, and cfish (See *.BSS)
- ? => print this help list and stop (also see -h)

For example, the command line

```
C:\BASS22> bass_v22 -i input.prj -l -c
```

will execute the project file INPUT.PRJ without simulating acute or chronic chemical lethality and report the distribution of cpu time spent within various key BASS subroutines.

Table 4.1 Valid Unit Prefixes.

Prefix Name	Conversion
	Factor
atto	10^{-18}
centi	10^{-02}
deca	10^{-01}
deci	10^{-01}
exa	10^{+18}
femto	10^{-15}
giga	10^{+09}
hecto	10^{-02}
kilo	10^{+03}
mega	10^{+06}
micro	10^{-06}
milli	10^{-03}
myria	10^{-04}
nano	10^{-09}
peta	10^{+15}
pico	10^{-12}
tera	10^{+12}

Table 4.2 Valid Unit Names for Length, Area, Volume, Mass, Time, and Energy. This list is not exhaustive and summarizes only commonly used unit names that BASS's units conversion program recognizes.

Unit Name	Conversion Factor to SI	Metre	Kg	Second	Description
acre	2.471×10^{-04}	2	0	0	4840 yards ²
are	1.000×10^{-02}	2	0	0	100 meter ²
btu	9.479×10^{-04}	2	1	-2	
calorie	2.388×10^{-01}	2	1	-2	
cc	$1.000 \times 10^{+06}$	3	0	0	cm ³
cm	$1.000 \times 10^{+02}$	1	0	0	
day	1.157×10^{-05}	0	0	1	
decade	3.169×10^{-09}	0	0	1	10 years
erg	$1.000 \times 10^{+07}$	2	1	-2	
fathom	5.468×10^{-01}	1	0	0	6 feet
feet	$3.281 \times 10^{+00}$	1	0	0	
foot	$3.281 \times 10^{+00}$	1	0	0	
ft	$3.281 \times 10^{+00}$	1	0	0	feet, foot
g	$1.000 \times 10^{+03}$	0	1	0	grams
gallon	$2.642 \times 10^{+02}$	3	0	0	3.785 liter
gm	$1.000 \times 10^{+03}$	0	1	0	grams
gram	$1.000 \times 10^{+03}$	0	1	0	
gramme	$1.000 \times 10^{+03}$	0	1	0	
hectare	1.000×10^{-04}	2	0	0	100 are
hour	2.778×10^{-04}	0	0	1	
hr	2.778×10^{-04}	0	0	1	hour
imperialgallon	$2.200 \times 10^{+02}$	3	0	0	4.54 liter
inch	$3.937 \times 10^{+01}$	1	0	0	
joule	$1.000 \times 10^{+00}$	2	1	-2	
kg	$1.000 \times 10^{+00}$	0	1	0	kilograms
km	1.000×10^{-03}	1	0	0	kilometer
l	$1.000 \times 10^{+03}$	3	0	0	liter
lb	$2.205 \times 10^{+00}$	0	1	0	pound
liter	$1.000 \times 10^{+03}$	3	0	0	
litre	$1.000 \times 10^{+03}$	3	0	0	
m	$1.000 \times 10^{+00}$	1	0	0	meter
meter	$1.000 \times 10^{+00}$	1	0	0	
metre	$1.000 \times 10^{+00}$	1	0	0	
mg	$1.000 \times 10^{+06}$	0	1	0	milligrams
micron	$1.000 \times 10^{+06}$	1	0	0	10 ⁻⁶ meter
mile	6.214×10^{-04}	1	0	0	5280 feet
min	1.667×10^{-02}	0	0	1	minute
minute	1.667×10^{-02}	0	0	1	
ml	$1.000 \times 10^{+06}$	3	0	0	
mm	$1.000 \times 10^{+03}$	1	0	0	

Table 4.3 Valid Ecological, Morphometric, and Physiological Unit Names.

Unit Name	Conversion Factor to SI	Metre	Kg	Second	Description
fish	n.a.	0	0	0	treated as information as is byte
g(O2)	7.3718×10^{-5}	2	1	-2	gram of oxygen
mg(O2)	7.3718×10^{-2}	2	1	-2	milligram of oxygen
ug(O2)	7.3718×10	2	1	-2	microgram of oxygen
ha	1.000×10^{-4}	2	0	0	hectare
individuals	n.a.	0	0	0	treated as information as is byte
inds	n.a.	0	0	0	treated as information as is byte
kcal	2.388×10^{-4}	2	1	-2	kilocalorie
ul(O2)	5.1603×10	2	1	-2	microliter oxygen STP = micromole
ml(O2)	5.1603×10^{-2}	2	1	-2	milliliter oxygen STP = millimole
l(O2)	5.1603×10^{-5}	2	1	-2	22.4 liters STP = mole
lamellae	n.a.	0	0	0	treated as information as is byte
umol(O2)	2.3037	2	1	-2	micromole of oxygen
mmol(O2)	2.3037×10^{-3}	2	1	-2	millimole of oxygen
mol(O2)	2.3037×10^{-6}	2	1	-2	mole of oxygen

Note: For purposes of units conversion, all units for oxygen consumption are treated dimensionally as joules.

5. BASS Model Software and Graphical User Interface

5.1. Software Overview

The BASS v2.2 model and Graphical User's Interface (GUI) software are provided via two downloads from the USEPA Center for Exposure Assessment Modeling (CEAM) website (<http://www.epa.gov/ceampubl/>). These downloads are:

1. **Install_BASS_v22.exe**: An InstallShield executable file that installs the BASS model software, user's manual, distribution examples, GUI, and Output Analyzer for Windows NT/2K/XP.
2. **Install_BASS_v22_ModelOnly.exe**: A WinZip Self-extracting file that installs only the BASS model software, user's manual, and distribution examples for all DOS/Windows systems.

The installation program **Install_BASSv22.exe** creates and installs the BASS model software, GUI, and Output Analyzer into the installation directory shown below

```
PATH \BASS_V22
      BASS_V22.EXE
      BASS_V22_ABSOFT.EXE
      BASS_V22_LAHEY.EXE
      DISDLL.DLL
      CLEAN_EXAMPLES.BAT
      RUN_EXAMPLES.BAT
      \BASS_CMM_FSH
        BASS_CMM_FSH.EXE
        BASS_FISH_CODES.DB
        BASS_FISHPAR.DB
        \SOURCE_CODE
      \BASS_GUI
      \COMMUNITY
      \DOCUMENTS
        BASS_MANUAL.WPD
        BASS_MANUAL.PDF
        BASS_DATA_SUPPLEMENT.WPD
        BASS_DATA_SUPPLEMENT.PDF
      \FISH
      \PROPERTY
      \PROJECTS
        \EX_EVERGLADES_CANAL
        \EX_EVERGLADES_CANAL_FISHING
        \EX_EVERGLADES_CANAL_HG
        \EX_EVERGLADES_CANAL_LESLIE_HG
        \EX_EVERGLADES_HOLES_HG
        \EX_EVERGLADES_MARSH_HG
        \EX_L_HARTWELL
        \EX_L_HARTWELL_PCB
        \EX_L_HARTWELL_PCB_TRANS
        \EX_L_ONTARIO_PCB
        \EX_SE_FARM_POND
      \SOURCE_CODE
```

where *PATH* = C:\PROGRAM FILES\BASS unless changed by the

user. The contents of this directory are:

1. **BASS_V22_ABSOFT.EXE** is the most current BASS model executable that has been created with the Absoft MP version 9.0 Fortran 95 compiler using the standard 32-byte Windows XP operating system running on an IBM IntelliStation A Pro equipped with dual 64-byte Opteron processors. On single processor machines, this executable will run approximately 3 times faster than the Lahey-Fujitsu executable; on dual processor machines, this executable will run approximately 5-7 times faster than the Lahey-Fujitsu executable. Note, however, that there are minor compiler bugs associated with this executable that apparently do not affect its computational accuracy. Also see Section 7.2.7.
2. **BASS_V22_LAHEY.EXE** is the most current BASS model executable that has been created with the Lahey-Fujitsu Fortran 95 version 5.7f compiler. This executable is used as the default BASS software executable **BASS_V22.EXE**.
3. **DISDLL.DLL** is a dynamic link library needed to execute the DISLIN graphing software.
4. **\BASS_GUI** contains the executables and associated library and support files for the BASS GUI and Output Analyzer.
5. **\BASS_CMM_FSH** contains the BASS fish file and community file generator, described in Section 5.6, with its associated database files and source code.
6. **\COMMUNITY** is the folder designed to be a repository of community files (*.CMM) that the user wishes to save as a canonical library for the construction of future BASS projects. Although this folder is empty, it must be present for the BASS software to function correctly. See Section 4.5 (page 56).
7. **\DOCUMENTS\BASS_MANUAL.WPD** is the current BASS User's Manual in WordPerfect format.
8. **\DOCUMENTS\BASS_MANUAL.PDF** is the current BASS User's Manual in PDF format.
9. **\DOCUMENTS\BASS_V22_DATA_SUPPLEMENT.WPD** is the current compendium of fish data that can be used to parameterize BASS in WordPerfect format.
10. **\DOCUMENTS\BASS_V22_DATA_SUPPLEMENT.PDF** is the current compendium of fish data that can be used to parameterize BASS in PDF format.

11. **\FISH** is the folder designed to be a repository of fish files (*.FSH) that the user wishes to save as a canonical library for the construction of future BASS projects. Although this folder is empty, it must be present for the BASS software to function correctly. See Section 4.5 (page 56).

12. **\PROJECTS** contains the BASS v2.2 distribution example projects that are described in Section 6.1 (page 75). All of these examples can be executed by double-clicking on the DOS batch file **RUN_EXAMPLES.BAT**.

13. **\PROPERTY** is the folder designed to be a repository of chemical property files (*.PRP) that the user wishes to save as a canonical library for the construction of future BASS projects. This folder must be present for the BASS software to function correctly, and it is initially populated with chemical property files used by the BASS distribution examples. This folder also contains the folder **\BARBER_2003** which contains chemical property files for the chemicals analyzed in Barber's review paper of gill exchange models (Barber, M.C. 2003. Environ. Toxicol. Chem. 22: 1963-1992). See Section 4.5 (page 56).

14. **\SOURCE_CODE** contains the current Fortran 95 source code for BASS v2.2. This folder is included for those users who would like to review the BASS code or to adapt it for other purposes.

The installation program **Install_BASS_v22_ModelOnly.exe** extracts a copy of the aforementioned installation directory **BASS_v22** that does not include the **\BASS GUI** subdirectory.

5.2. Installation Notes

The BASS model and GUI v2.2 has been installed and successfully tested on systems running Win2000, WinNT4.0 and WinXP operating systems with various configurations of each. If users are running NT, 2000, or XP operating systems, they must have Administrator privileges on their systems in order to install the BASS model and GUI software.

5.3. Installation Procedures

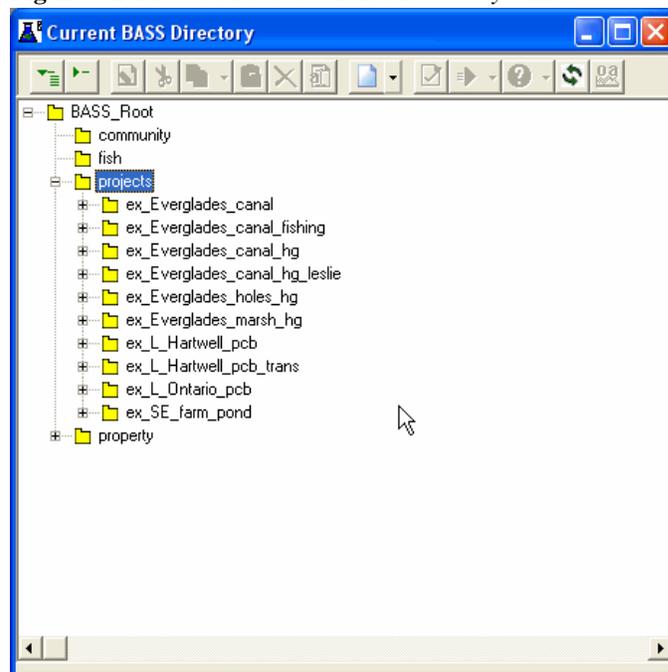
For complete installation procedures users are referred to the BASS installation readme file at the USEPA Center for Exposure Assessment Modeling (CEAM) website (<http://www.epa.gov/ceampubl/>).

5.4. BASS GUI Operation

The BASS GUI has been designed to emulate Microsoft's Windows Explorer in much of its form and function. After the BASS GUI is opened, the first window that users see is the GUI's

Current BASS Directory (see **Figure 5.1**). If this window is inadvertently closed, it can be reopened using the *View* button found on the toolbar of the GUI's host window *BASS version 2.2*

Figure 5.1 BASS GUI *Current BASS Directory* window.



Double-clicking on a folder's name, icon, or directory node expands or collapses the folder's contents into or out of the user's view, respectively. Double-clicking on a file name opens the file with one of six GUI file editors based on the selected file's extension. The GUI's file editors can also be invoked by:

1. Left-clicking on the file and pressing the *Enter* key.
2. Right-clicking on the file and then left-clicking on *Edit*.
3. Left-clicking on the file and left-clicking on the *Edit* icon  found on the *Current BASS Directory* toolbar.

When users are editing a BASS project file that contains include files, users can also open file editors for those include files by

4. Left-clicking on the desired include command and then left-clicking on the resulting activated *Open Include File* link (see Section 5.4.1).

BASS output files (i.e., *.BSS, *.MSG, and *.XML), are not displayed in the *Current BASS Directory* window. These files, if they exist, are accessed via the project files (*.PRJ) that generated them.

BASS message files (*.MSG) and simulation summary files (*.BSS) can be reviewed by right-clicking on the relevant project file and then left-clicking on *View Project Message File* or *View BSS File*, respectively. These files can also be reviewed by left-clicking on the desired project file and then left-clicking on the arrow of the *File Viewing* icon  found on the *Current BASS Directory* toolbar. The *File Viewing* icon has an associated drop-down selection that enables users to specify which output file type is to be viewed. If the *File Viewing* icon is left-clicked directly, the project's message file is opened by default.

BASS XML files can be loaded into the BASS Output Analyzer either by right-clicking on the relevant project file and then left-clicking on *View Output Analyzer* or by left-clicking on the desired project file and then left-clicking on the *Plotting* icon  found on the *Current BASS Directory* toolbar.

BASS project files are executed either by right-clicking on the desired project file and then left-clicking on *Run Project* or by left-clicking on the desired project file and then left-clicking on the arrow of the *Execution* icon  on the *Current BASS Directory* toolbar. Like the *File Viewing* icon, the *Execution* icon has an associated drop-down selection that enables users to specify command line options as described in Section 4.7 (page 56). When a project file is being executed, all other GUI functions are unavailable until the simulation is completed.

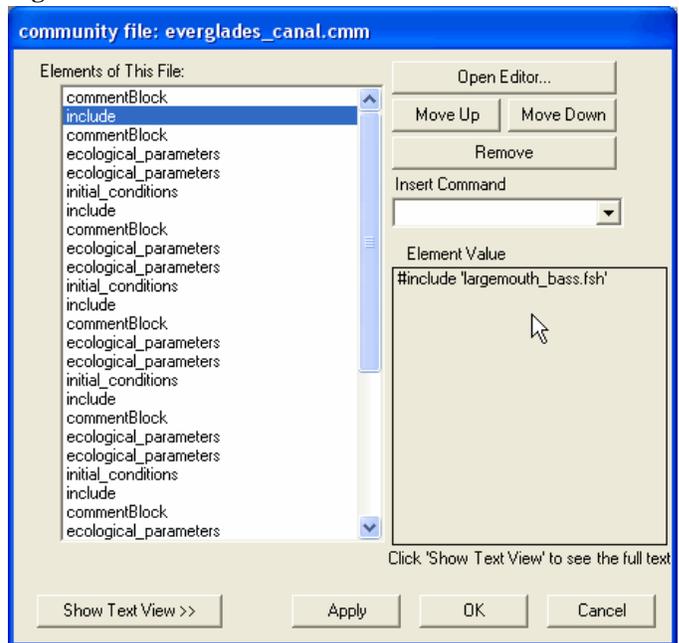
BASS project files can be checked for their syntax and data completeness before attempting execution either by right-clicking on the desired project file and then left-clicking on *Validate Project* or by left-clicking on the desired project file and then left-clicking on the *Validate Project* icon  on the *Current BASS Directory* toolbar. If the project file has syntax errors or missing input data, the GUI's *Event Viewer* will automatically open and display validation status of the project as well as associated errors and warnings. Most users, however, will find it easier to review these errors by opening the project's MSG file, as outlined previously, and search for the phrase "ERROR:" to determine the needed corrective actions.

5.4.1. BASS File Editors

All six GUI file editors have the same essential format and function as displayed in **Figure 5.2**. Commands, include files, and comment blocks contained within the file being edited are displayed in abbreviated form and in order of their appearance within the *Elements of This File* box. The full details of these elements can be viewed individually within the *Element Value* box or as they appear within the file by left-clicking on the *Show Text View* toggle button. Elements can be edited by either

double-clicking on the element name or by left-clicking on the element and then left-clicking on the *Open Editor...* button.

Figure 5.2 General structure of BASS GUI file editors.

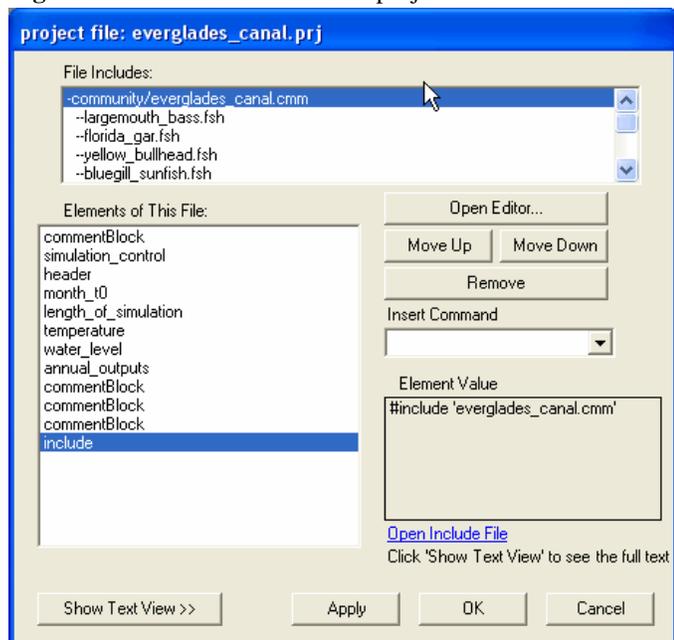


The position of elements can be changed by using the *Move Up* and *Move Down* buttons. Existing elements can be removed and new elements added by using the *Remove* button and *Insert Command* box, respectively. When elements are either added, removed, or reordered, however, users must first left-click on the *Apply* button before opening any GUI command editor. The *Apply* button is also used to save editorial changes at any time during an editing session.

Because the typical *Close* "X" button has been disabled on all GUI file editors, users can exit GUI file editors only by using the *OK* and *Cancel* buttons. These buttons either save or cancel any editorial changes since the last invocation of the *Apply* button. This GUI behavior is designed to preserve the integrity of the GUI's Document Object Model (DOM).

Figure 5.3 displays the structure of the BASS GUI project file editor. This editor differs from the GUI's other five file editors in two ways. First, this editor explicitly identifies all include files that will be used by the project. Secondly, any include file that is directly referenced by the project file can be opened and edited by left-clicking on the *Open Include File* hyperlink that appears below the *Element Value* box whenever an include statement is highlighted in the *Elements of This File* box.

Figure 5.3 Structure of BASS GUI project file editor.



5.4.2. BASS Command Editors

GUI command editors are opened from GUI file editors as outlined in Section 5.4.1. In terms of their appearance and functionality, there are 17 basic command editor types that are described in the following:

- Simple String Editors that edit the commands /CHEMICAL, /COMMON_NAME, /HEADER, /SPECIES, and include file specifications (i.e., #INCLUDES . . .). See **Figure 5.4**.
- Simple String Editor with pull-down selection that edits the commands /AGE_CLASS_DURATION and /MONTH_T0. See **Figure 5.5**.
- Numeric Editor with units that edits the command /LENGTH_OF_SIMULATION. See **Figure 5.6**.
- Numeric Editor without units that edits the commands /ANNUAL_OUTPUTS, /LOG_AC, /LOG_KB1, /LOG_KB2, /LOG_P, /MELTING_POINT, /MOLAR_VOLUME, and /MOLAR_WEIGHT. See **Figure 5.7**.
- Forcing Function Editor that edits the commands /BIOTA, /EXPOSURE, /HABITAT_PARAMETERS, /TEMPERATURE, and /WATER_LEVEL. See **Figure 5.8**.
- Feeding Model Editor that edits the command /FEEDING_OPTIONS. See **Figure 5.9**.

- Compositional and Morphometric Editor that edits the commands /COMPOSITIONAL_PARAMETERS and /MORPHOMETRIC_PARAMETERS. See **Figure 5.10**.
- Ecological Editor that edits the command /ECOLOGICAL_PARAMETERS. See **Figure 5.11** and **Figure 5.12**.
- Physiological and Growth Editor that edits the command /PHYSIOLOGICAL_PARAMETERS. See **Figure 5.13**.
- Cohort Initial Conditions Editor that edits the command /INITIAL_CONDITIONS. See **Figure 5.14**.
- Spawning Period Editor that edits the command /SPAWNING_PERIOD. See **Figure 5.15**.
- Fishery Editor that edits the command /FISHERY_PARAMETERS. See **Figure 5.16**.
- Nonfish Biotic Editor that edits the commands /BENTHOS, /PERIPHYTON, /PHYTOPLANKTON, /TERRESTRIAL_INSECTS, and /ZOOPLANKTON. See **Figure 5.17** and **Figure 5.18**.
- Nonfish BCF Editor that edits the command /NONFISH_BCF. See **Figure 5.19**.
- Chemical Metabolism Editor that edits the command /METABOLISM. See **Figure 5.20**.
- Chemical Toxicity Editor that edits the command /LETHALITY. See **Figure 5.21**.
- Plot Selection Editor that edits the commands /ANNUAL_PLOTS and /SUMMARY_PLOTS. See **Figure 5.22**.

As noted with the GUI file editors, the typical *Close* “X” button has been disabled on all GUI command editors. Users can only exit or close a command editor by using the *OK* and *Cancel* buttons. These buttons either save or cancel any editorial changes since the editor was opened. This GUI behavior is designed to preserve the integrity of the GUI’s Document Object Model (DOM).

5.4.3. Special Function Editors

In addition to the file and command editors described in the previous section, the BASS GUI has two special function editors, i.e.,

- Comment Block Editor that is used to insert comment blocks before or after BASS commands, as opposed to end-of-line

comments associated with the individual options of BASS commands. See **Figure 5.23**.

- Time Series Data Editor for editing external data files that are specified as file functions (e.g., /BIOTA, /EXPOSURE, /HABITAT_PARAMETERS, /TEMPERATURE, and /WATER_LEVEL). See **Figure 5.24**.

5.4.4. File and Folder Operations

Using the GUI's *Current BASS Directory* window, users can create new files and project folders either from scratch or from existing files and project folders.

To create a BASS project or include file from scratch, users must first left-click on the subdirectory (i.e., \COMMUNIY, \FISH, or \PROPERTY) or project folder where the file is to be created. The user then must left-click on the drop-down arrow head of the *Add New File* icon . When the *Add New File* drop-down menu appears, the user must left-click on the desired file type to be created. Finally, after the new file appears in the *Current BASS Directory* window, the user must complete the naming of the new file. New project folders can be created following these same steps.

Users can create a file from an existing file by

1. Left-clicking on the desired file and then left-clicking on the *Copy* icon .
2. Left-clicking on the desired destination folder or subdirectory and left-clicking on the *Paste* icon .

Users can also create a new file from an existing file by

1. Right-clicking on the file to be copied and then left-clicking on *Copy*.
2. Right-clicking on the destination folder or subdirectory and then left-clicking on *Paste*.

Lastly users can create a new file from an existing file by

1. Left-clicking on the file to be copied and then pressing CTRL-c.
2. Left-clicking on the destination folder or subdirectory and then pressing CTRL-v

New project folders can be created from existing projects using the same procedures.

5.5. The BASS Output Analyzer

The BASS Output Analyzer (OA) is a dual purpose post-processor that enables users to construct customized graphs and tables. This software can be invoked either from within the BASS GUI or as a standalone application. Using this software, users can create two and three-dimensional graphs of any state variable that is a valid option for the plotting commands /ANNUAL_PLOTS or /SUMMARY_PLOTS. Unlike the plots generated by these commands, however, users can generate plots for only selected species as desired. Additionally, users can specify arbitrary time periods of interest as well as change the number of size or age classes that are plotted. The BASS OA also enables users to create customized versions of the summary tables that are generated for BSS output files. These tables can be copied and pasted into either Word or WordPerfect documents.

When BASS executes a user's project file, two XML files are generated for use by the BASS OA. Both of these files reside in the project folder of the PRJ file that generated them. The first of these files contains the actual data that the OA will use for graphing and table construction. This file bears the same name as its associated project file but possesses the extension XML. Importantly, it is this file that users must open when using the BASS OA.

The second XML file generated for any particular BASS simulation contains general summary statistics of the simulation and is loaded automatically into the OA when the aforementioned XML file is opened. The name of this file is the associated project file's name appended with the string *_incl*.

5.6. Auxiliary BASS Software

To aid users in constructing BASS fish files and community files, an auxiliary piece of software named BASS_CMM_FSH.EXE is distributed with the BASS model and GUI software. Using a combination of an internal database of fish growth rates and two external database files (BASS_FISH_CODES.DB and BASS_FISHPAR.DB), this software can generate default FSH files for many North American freshwater fish. Although these generated FSH files are setup to use BASS's linear feeding model, users can easily edit these files to use BASS's allometric feeding model or a combination of both. Users can also construct multiple FSH files and an associated, rudimentary CMM file for an arbitrary selection of fish using this software. This software, however, does not have a GUI and must be executed by the user from a DOS command prompt.

To generate a FSH file for a single species of interest, the user should open a DOS command prompt window and navigate to the project folder in which they want the file to be generated. Assuming that the user's BASS root directory is C:\BASS_V22, the DOS command

```
...>C:\BASS_V22\BASS_CMM_FSH\bass_cmm_fsh.exe -f
"bluegill" -g "lepomis macrochirus" -m "January 10" -l 10 -u
33
```

will generate a fish file for bluegill sunfish whose growth rate has been calibrated to an annual sinusoidal water temperature cycle that varies from 10 to 33 Celsius and whose minimum annual temperature occurs on January 10.

To generate multiple FSH files and an associated CMM file for an arbitrary selection of fish, the user should again open a DOS command prompt window and navigate to the project folder in which the user wants the files to be generated. Assuming that the user's BASS root directory is C:\BASS_V22, the DOS command

```
...>C:\BASS_V22\BASS_CMM_FSH\bass_cmm_fsh.exe -i
fishes.dat0
```

will generate a FSH file for each fish species identified in the file *fishes.dat0* and an associated CMM file. The file *fishes.dat0* must reside in the desired project folder. An example of the general structure of these input files is illustrated below

```
! File:bass_bluegill_catfish.dat0

CMM_FILE_NAME bass_bluegill_catfish.cmm
MONTH_TO August
COLDEST_DAY January 10
TEMPERATURE_MAXIMUM 30
TEMPERATURE_MINIMUM 10

FISH_START micropterus salmoides
COMMON_NAME largemouth bass
```

```
SPAWNING_PERIOD april-may
parameter_option_1; comment/reference
parameter_option_2; comment/reference
:
parameter_option_n; comment/reference
biomass[kg/ha]= number ! or density[fish/ha]= number
FISH_END

FISH_START Lepomis macrochirus
COMMON_NAME bluegill sunfish
SPAWNING_PERIOD april-october
parameter_option_1; comment/reference
parameter_option_2; comment/reference
:
parameter_option_n; comment/reference
biomass[kg/ha]= number ! or density[fish/ha]= number
FISH_END

FISH_START ictalurus punctatus
COMMON_NAME channel catfish
SPAWNING_PERIOD may-june
parameter_option_1; comment/reference
parameter_option_2; comment/reference
:
parameter_option_n; comment/reference
biomass[kg/ha]= number ! or density[fish/ha]= number
FISH_END
```

where *parameter_option_i* is any valid option for the BASS fish commands \COMPOSITIONAL_PARAMETERS, \ECOLOGICAL_PARAMETERS, \MORPHOLOGICAL_PARAMETERS, or \PHYSIOLOGICAL_PARAMETERS that the user wants to supercede the default assignment made by BASS_CMM_FSH.EXE. Most of the FSH files used by the example BASS distribution projects have been generated using earlier versions of this software.

Figure 5.4 GUI command editor for simple strings.

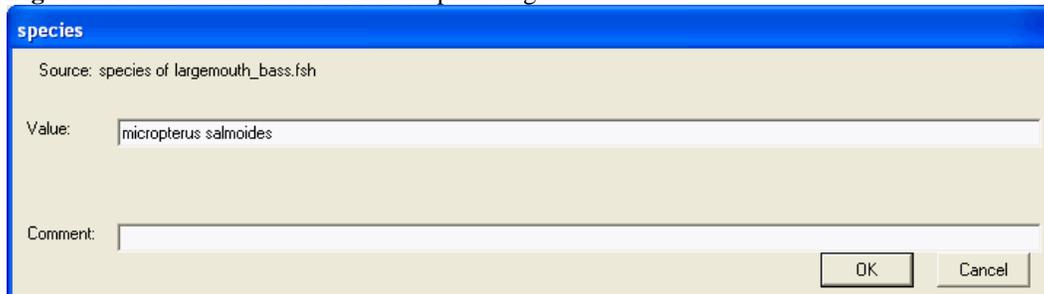


Figure 5.5 GUI command editor for simple strings with drop-down selection.

The GUI window is titled "age_class_duration". It shows the source as "age_class_duration of largemouth_bass.fsh". The "Value:" field is a drop-down menu currently showing "year". The "Comment:" field is an empty text box. "OK" and "Cancel" buttons are at the bottom right.

Figure 5.6 GUI command editor for numeric data with user specified units.

The GUI window is titled "length_of_simulation". It shows the source as "length_of_simulation of everglades.prj". The "Comment:" field is an empty text box. Below it is a table with the following data:

Parameter	Units	Value	Comments
length_of_simulation	year	20	

"OK" and "Cancel" buttons are at the bottom right.

Figure 5.7 GUI command editor for numeric data fixed units.

The GUI window is titled "annual_outputs". It shows the source as "annual_outputs of everglades.prj". The "Value:" field is a text box containing "20". The "Comment:" field is an empty text box. "OK" and "Cancel" buttons are at the bottom right.

Figure 5.8 GUI command editor for forcing functions.

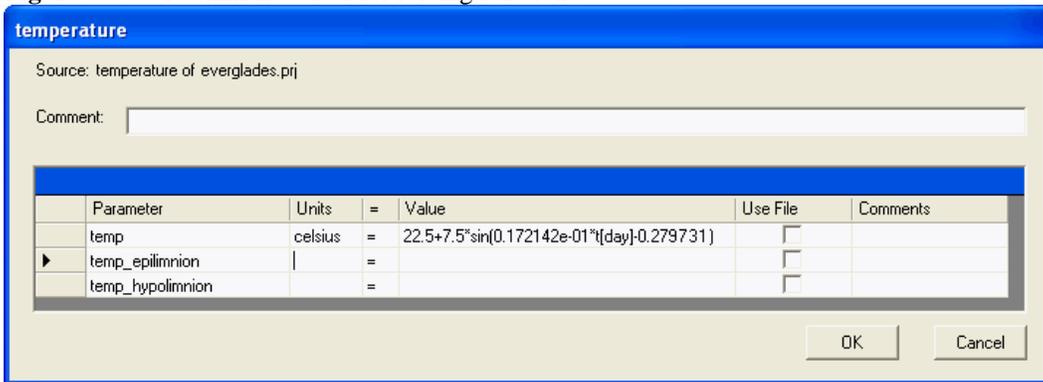


Figure 5.9 GUI command editor for feeding model options.

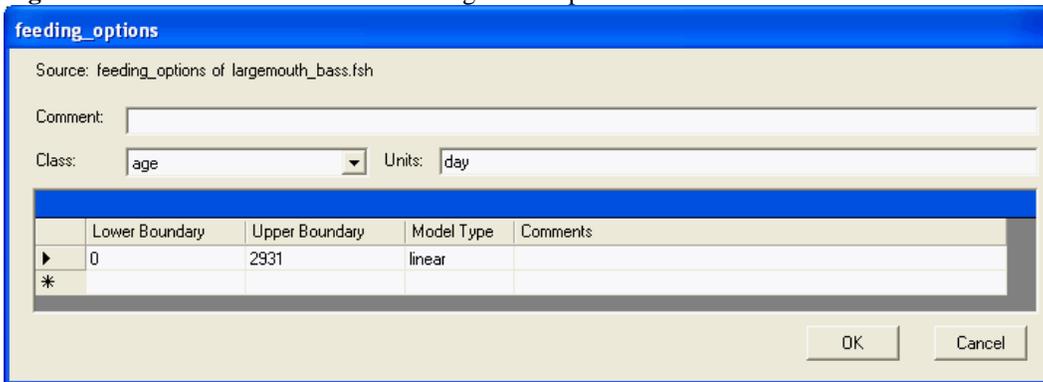


Figure 5.10 GUI command editor for compositional and morphometric parameters.

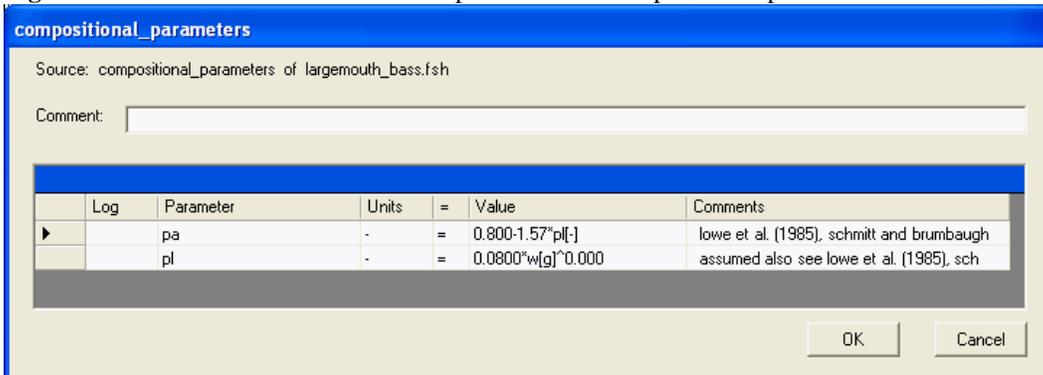


Figure 5.11 GUI command editor for nondiet ecological parameters.

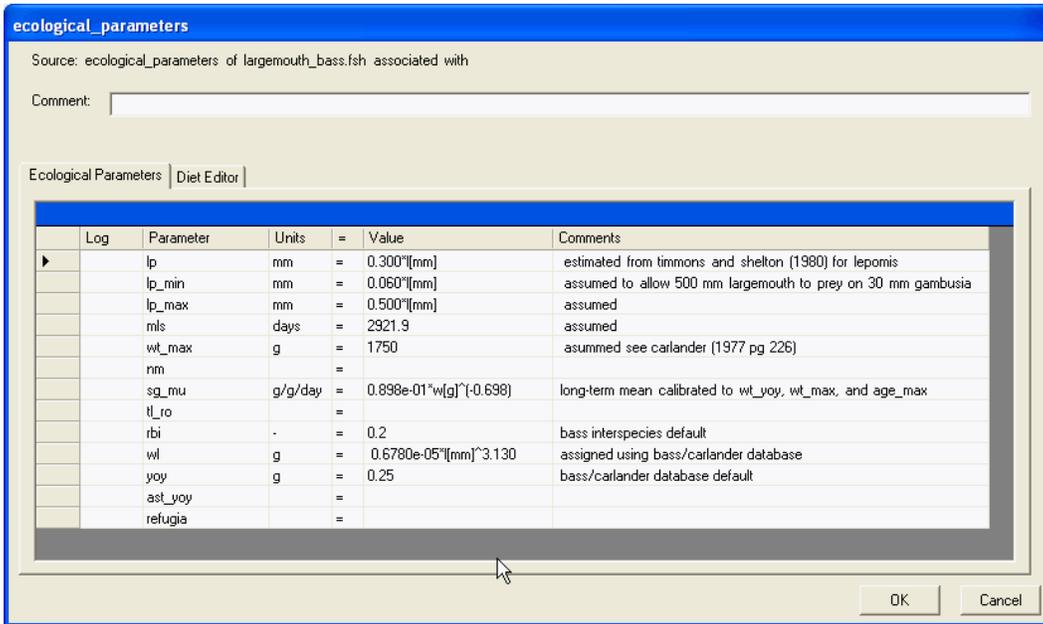


Figure 5.12 GUI command editor for fish diets.

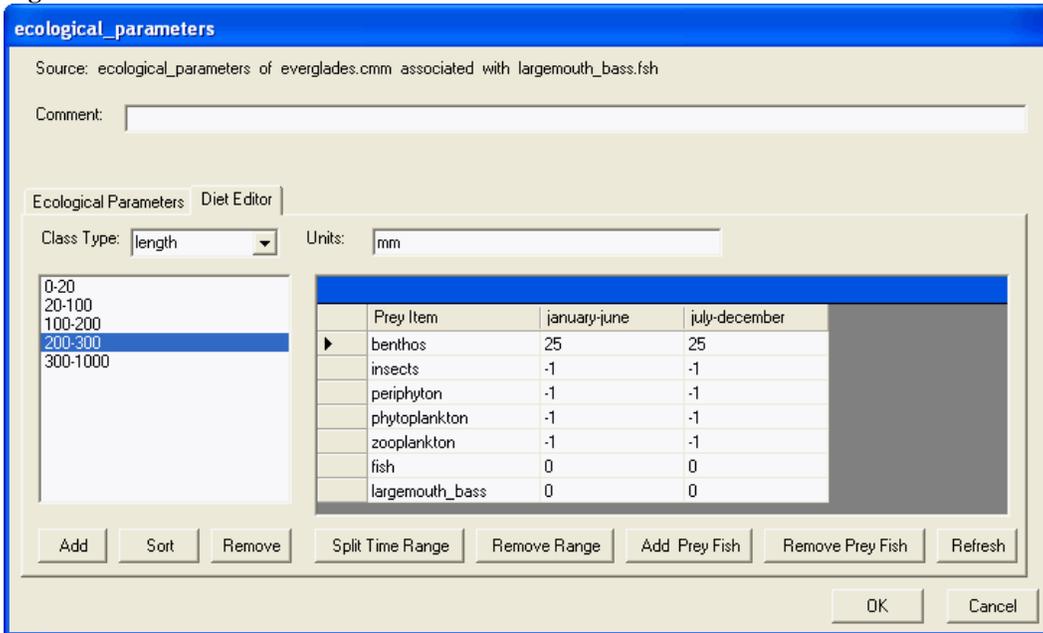


Figure 5.13 GUI command editor for physiological parameters.

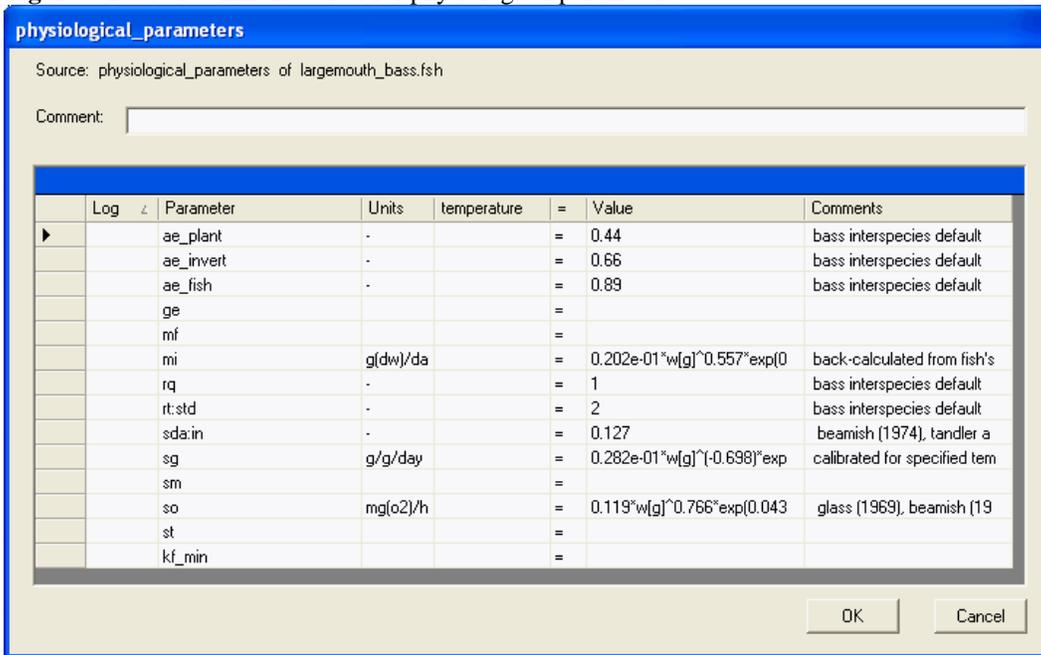


Figure 5.14 GUI command editor for cohort initial conditions.

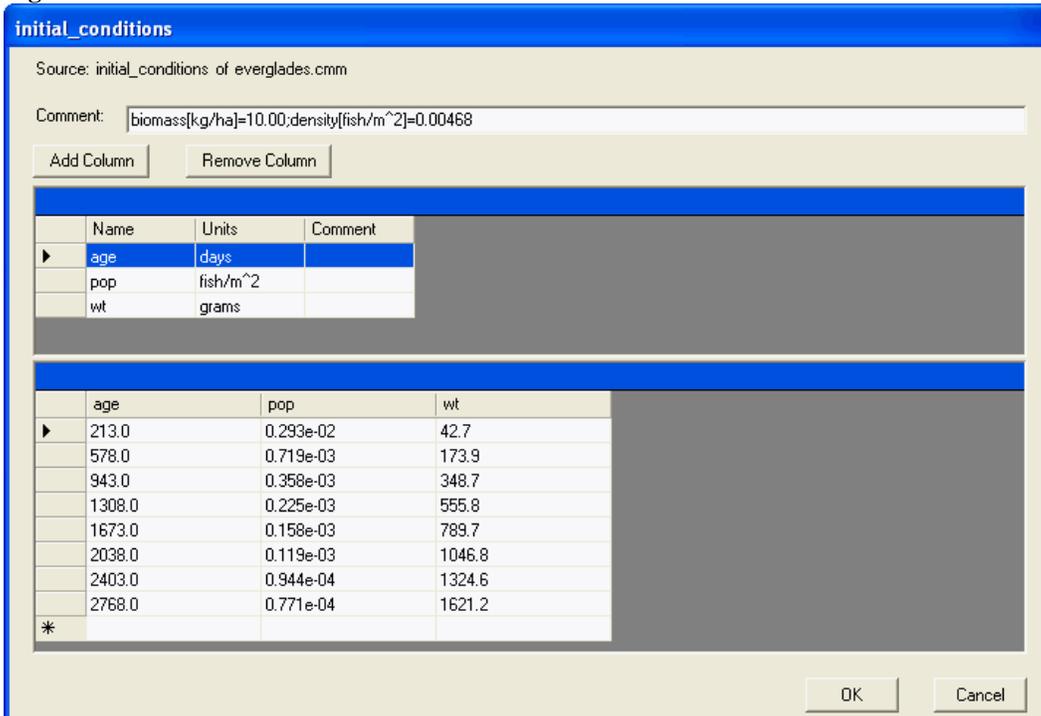


Figure 5.15 GUI command editor for spawning parameters.

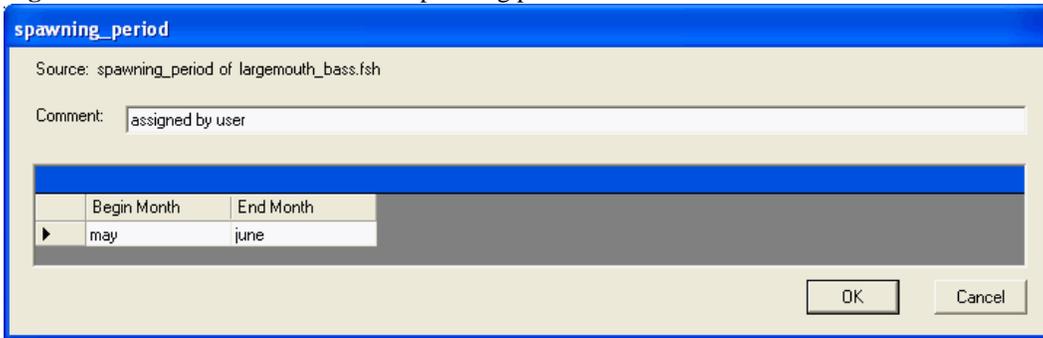


Figure 5.16 GUI command editor for fishery parameters.

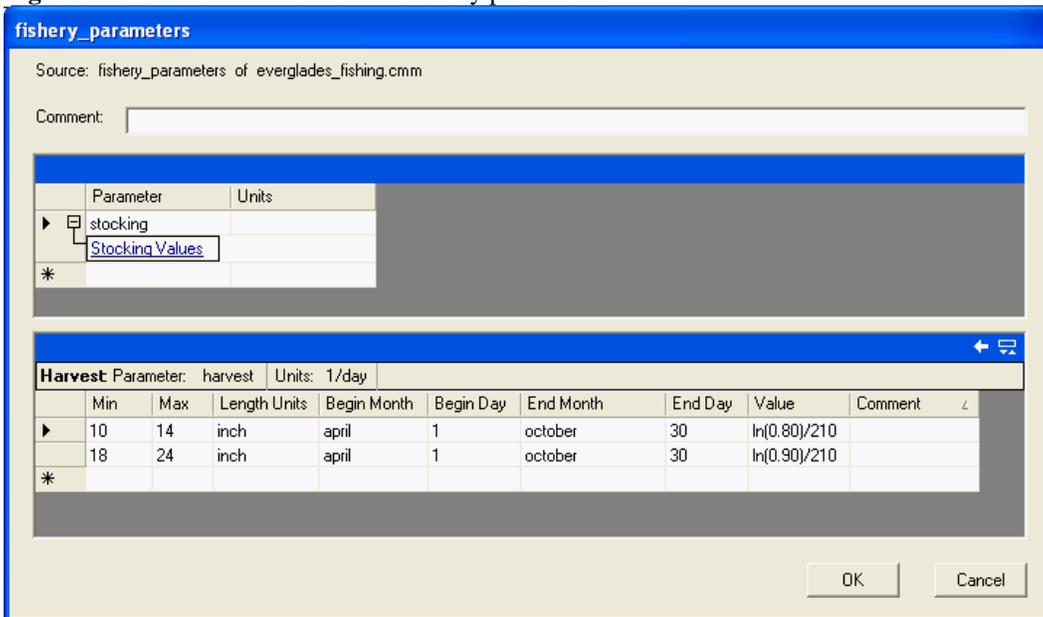


Figure 5.17 GUI command editor for nonfish biota as forcing functions.

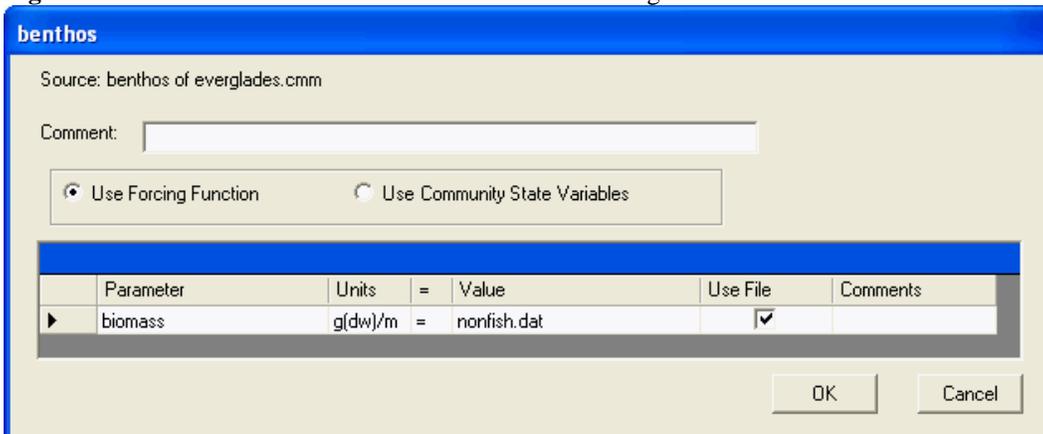


Figure 5.18 GUI command editor for nonfish biota as state variables.

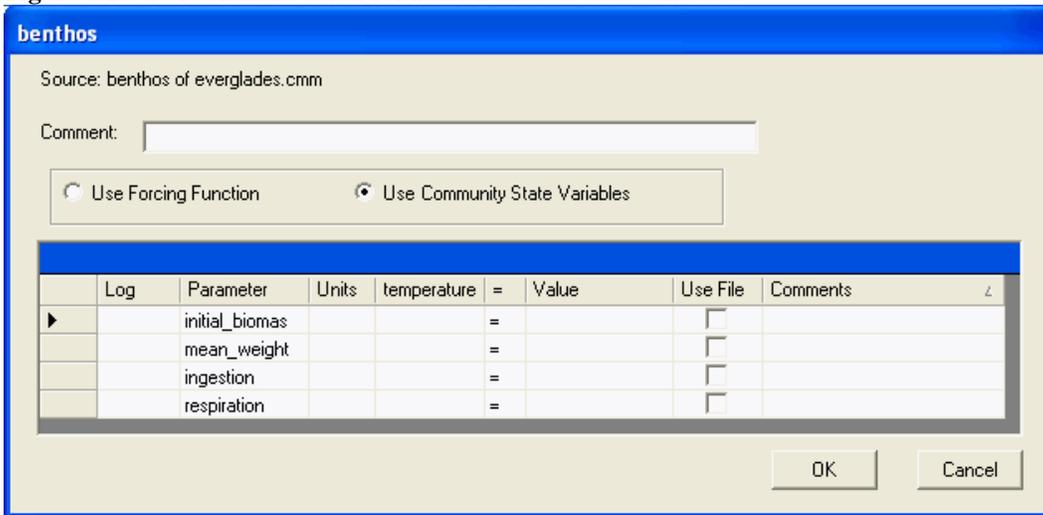


Figure 5.19 GUI command editor for nonfish bioaccumulation factors.

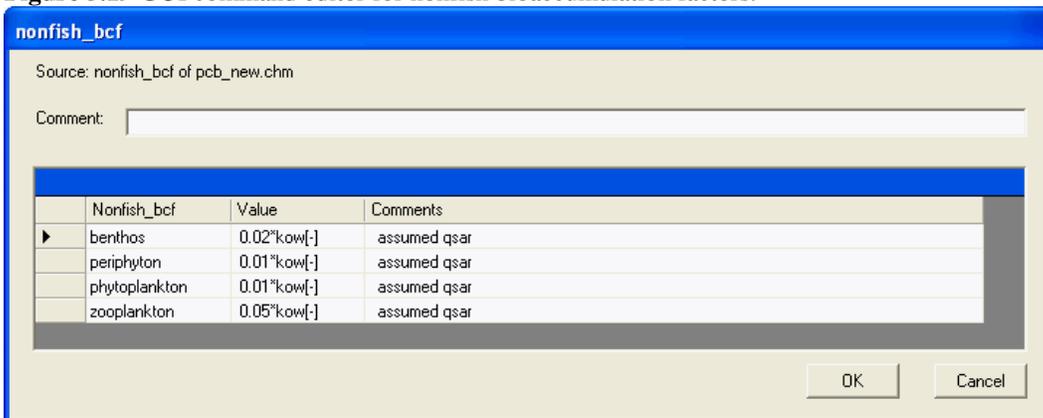


Figure 5.20 GUI command editor for chemical biotransformation parameters.

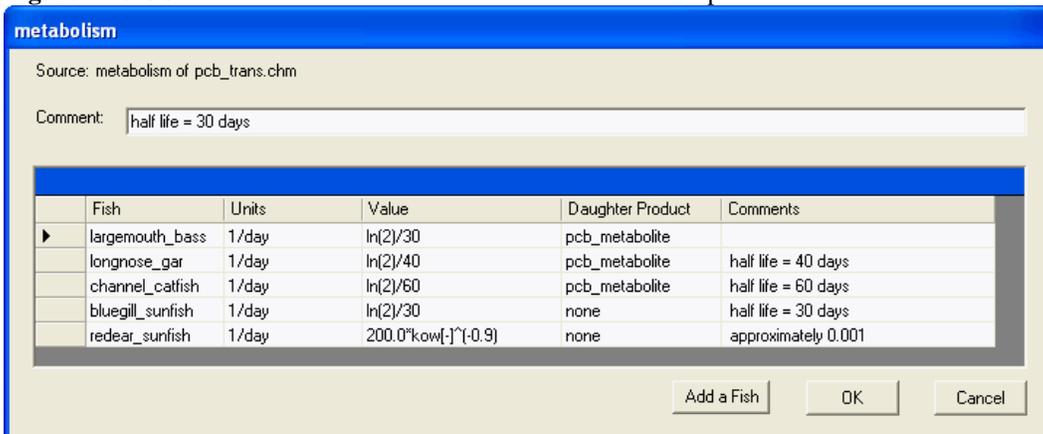


Figure 5.21 GUI command editor for chemical toxicity parameters.

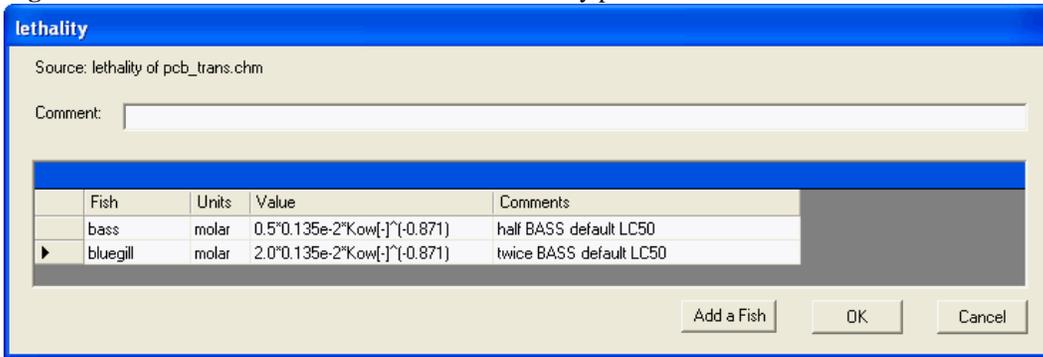


Figure 5.22 GUI command editor for automatic graphing selections.

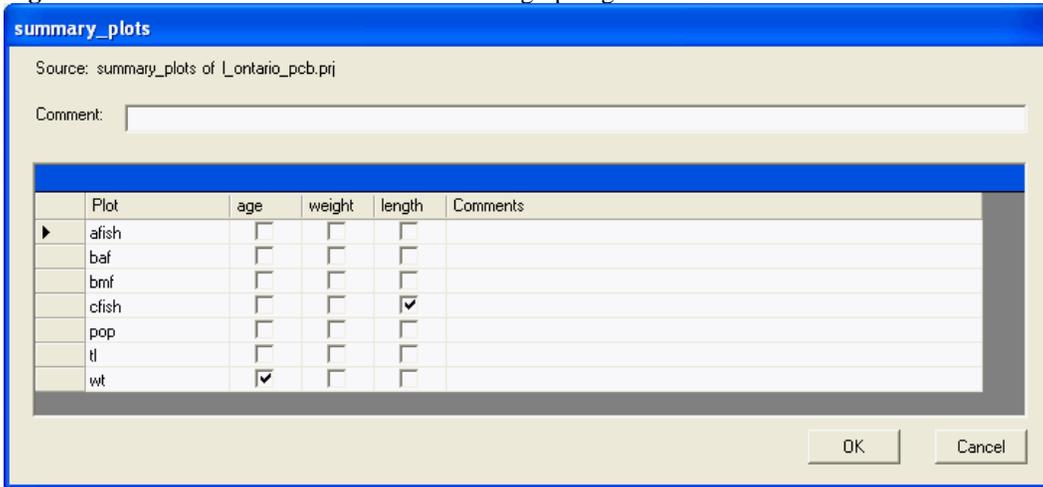


Figure 5.23 GUI Block comment editor.

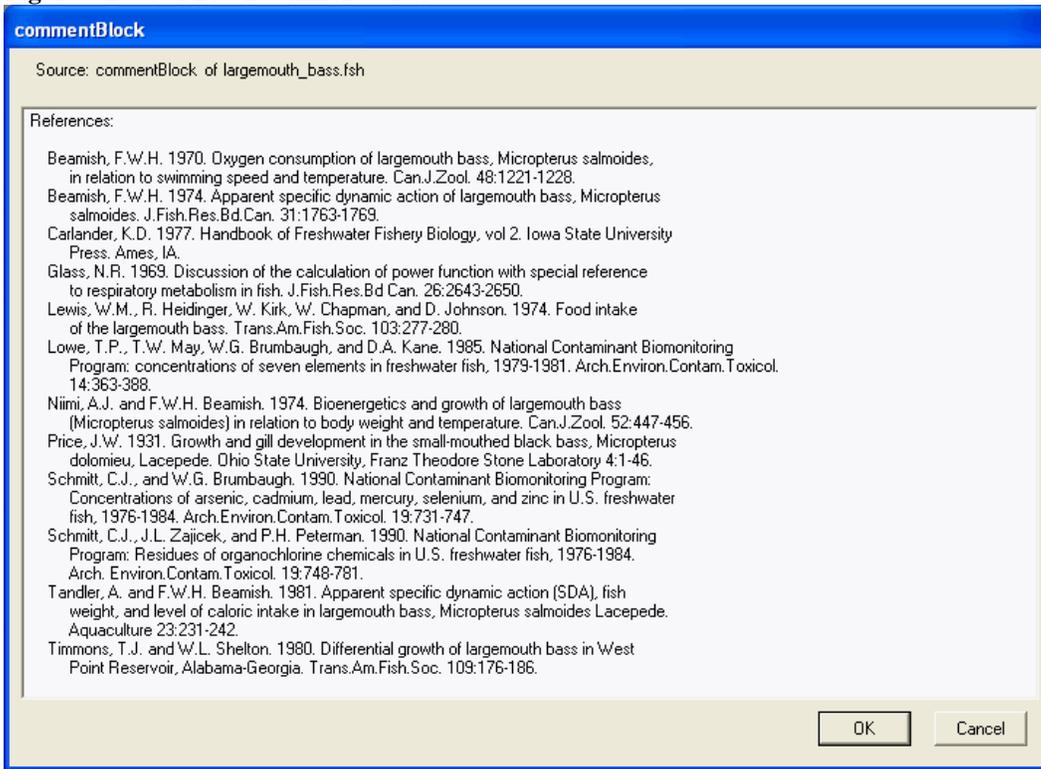
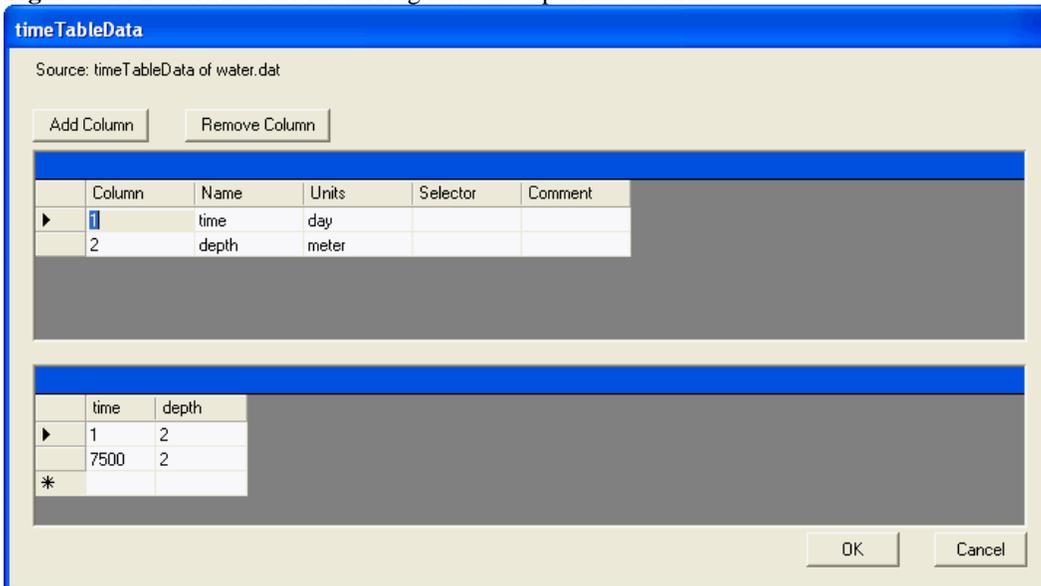


Figure 5.24 Data file editor for forcing functions specified as files.



6. Example Applications

6.1. BASS Software Distribution Examples

Several example projects are provided with the BASS model software and GUI. Each project resides in its own folder within the \PROJECTS subdirectory.

The example project EX_EVERGLADES_CANAL simulates the growth and population dynamics of a canal fish community in the Florida Everglades, USA using the project file EVERGLADES_CANAL.PRJ. The principal fish species in these communities are assumed to be bluegill sunfish (*Lepomis macrochirus*), Florida gar (*Lepisosteus platyrhincus*), largemouth bass (*Micropterus salmoides*), mosquito fish (*Gambusia holbrooki*), redeer sunfish (*Lepomis microlophus*), and yellow bullheads (*Ameiurus natalis*). The community file EVERGLADES_CANAL.CMM is used to specify the ecological and physiological parameters and the initial conditions for these species. Each species' daily consumption rate is back-calculated from its expected growth rate using BASS's linear feeding option. The community's water depth and the standing stocks of benthos, periphyton, and zooplankton are specified using the data files EVERGLADES_CANAL_WATER.DAT and EVERGLADES_NONFISH.DAT, respectively.

The example project EX_EVERGLADES_CANAL_FISHING simulates the growth and population dynamics of the aforementioned Everglades canal fish community assuming that largemouth bass, bluegill sunfish, and redeer sunfish are harvested by fishing. This example's project file EVERGLADES_CANAL_FISHING.PRJ uses the modified community file EVERGLADES_CANAL_FISHING.CMM to specify the fishes' ecological and physiological parameters, initial conditions, and assumed fishing mortalities. The community's water depth and non-fish standing stocks are again specified using the data files EVERGLADES_CANAL_WATER.DAT and EVERGLADES_NONFISH.DAT, respectively.

The example project EX_EVERGLADES_CANAL_HG simulates the bioaccumulation of methylmercury in an Everglades canal fish community using the project file EVERGLADES_CANAL_HG.PRJ. This example's project file uses the same community file as does the example project EVERGLADES_CANAL to specify the ecological and physiological parameters and initial conditions for the species of interest. The community's chemical exposures to methylmercury are provided by the include file EVERGLADES_MERCURY.CHM that in turn uses the include file \PROPERTY\METHYL_HG.PRP. As before, the community's water depth and the non-fish standing stocks are specified using the data files EVERGLADES_CANAL_WATER.DAT and

EVERGLADES_NONFISH.DAT, respectively.

The example project EX_EVERGLADES_CANAL_LESLIE_HG simulates the bioaccumulation of methylmercury in an Everglades canal fish community using BASS's Leslie matrix option and the project file EVERGLADES_CANAL_LESLIE_HG.PRJ. This project's ecological and physiological data are provided by the community file EVERGLADES_CANAL_LESLIE.CMM. Chemical exposures and properties of methylmercury are provided by the include file EVERGLADES_MERCURY.CHM that uses the include file \PROPERTY\METHYL_HG.PRP. Once again, the community's water depth and non-fish standing stocks are provided by the ancillary data files EVERGLADES_CANAL_WATER.DAT and EVERGLADES_NONFISH.DAT, respectively.

The example project EX_EVERGLADES_HOLES simulates the growth, population, and methylmercury dynamics of an alligator hole fish community in the Florida Everglades, USA using the project file EVERGLADES_HOLES.PRJ. The principal fish species in these communities are assumed to be bluegill sunfish, Florida gar, largemouth bass, least killifish (*Heterandria formosa*), mosquito fish, redeer sunfish, spotted sunfish (*Lepomis punctatus*), warmouth sunfish (*Lepomis gulosus*), and yellow bullheads. The community file EVERGLADES_HOLES.CMM is used to specify the ecological and physiological parameters and the initial conditions for these species. Each species' daily consumption rate is back-calculated from its expected growth rate using BASS's linear feeding option. The community's water depth and the standing stocks of benthos, periphyton, and zooplankton are specified by the project and community files EVERGLADES_HOLES_HG.PRJ and EVERGLADES_HOLES.CMM, respectively. Methylmercury exposures are provided by the include file EVERGLADES_MERCURY.CHM that uses the property file \PROPERTY\METHYL_HG.PRP as an include file.

The example project EX_EVERGLADES_MARSH simulates the growth, population, and methylmercury dynamics of an open marsh fish community in the Florida Everglades, USA using the project file EVERGLADES_MARSH.PRJ. The principal fish species in these communities are assumed to be bluefin killifish (*Lucania goodei*), Florida gar, golden top minnow (*Fundulus chrysotus*), largemouth bass, least killifish, mosquito fish, spotted sunfish, warmouth sunfish, and yellow bullheads. The community file EVERGLADES_MARSH.CMM is used to specify the ecological and physiological parameters and the initial conditions for these species. Each species' daily consumption rate is back-calculated from its expected growth rate using BASS's linear feeding option. The community's

water depth and the standing stocks of benthos, periphyton, and zooplankton are specified by the project and community files `EVERGLADES_MARSH_HG.PRJ` and `EVERGLADES_MARSH.CMM`, respectively. Methylmercury exposures are provided by the include file `EVERGLADES_MERCURY.CHM` that uses the property file `\PROPERTY\METHYL_HG.PRP` as an include file.

The example project `EX_L_HARTWELL_PCB` simulates the bioaccumulation of tetra-, penta-, hexa-, and hepta-PCB in a largemouth bass/ gizzard shad/ sunfish/catfish community in the Twelve Mile Creek arm of Lake Hartwell, SC, USA using the project file `TWELVEMILE_CREEK_PCB.PRJ`. The community's ecological and physiological parameters and initial conditions are specified using the community file `TWELVEMILE_CREEK.CMM`. Chemical exposures and properties of tetra-, penta-, hexa-, and hepta-PCBs are provided by the include files `TWELVEMILE_CREEK.CHM` `\PROPERTY\PCB_TETRA.PRP`, `\PROPERTY\PCB_PENTA.PRP`, `\PROPERTY\PCB_HEXA.PRP`, and `\PROPERTY\PCB_HEPTA.PRP`, respectively. This example demonstrates BASS's ability to simulate the bioaccumulation of arbitrary mixtures.

The example project `EX_L_HARTWELL_PCB_TRANS` also simulates the bioaccumulation of tetra-, penta-, hexa-, and hepta-PCB in the aforementioned largemouth bass/ gizzard shad/ sunfish/catfish community of Lake Hartwell, SC, USA using the project file `TWELVEMILE_CREEK_PCB_TRANS.PRJ`. This example, however, allows for the biotransformation of selected PCB congeners by selected fish species. The community file `TWELVEMILE_CREEK.CMM` is again used to specify the ecological and physiological parameters and the initial conditions for this community.

The example project `EX_L_ONTARIO_PCB` simulates the bioaccumulation of tetra-, penta-, hexa-, and hepta-PCB in Lake Ontario salmonids and alewife using BASS's "fgets" option and the project file `BARBER_ET_AL_1991.PRJ`. This example is the BASS implementation of the FGETS application published by Barber et al. (1991). Whereas salmonid feeding is simulated using BASS's Holling feeding option, the feeding by alewife is simulated using BASS's clearance feeding option. The community file `BARBER_ET_AL_1991.CMM` is used to specify the ecological and physiological parameters and the initial conditions for this community.

The example project `EX_SE_FARM_POND` simulates the growth and population dynamics of a typical southeastern US farm pond community using the project file `SE_FARM_POND.PRJ`. The principal fish species in these communities are assumed to be largemouth bass (*Micropterus salmoides*), bluegill sunfish (*Lepomis macrochirus*), redear sunfish (*Lepomis*

microlophus), redbreast sunfish (*Lepomis auritus*), channel catfish (*Ictalurus punctatus*), and yellow bullheads (*Ameiurus natalis*). The community file `SE_FARM_POND.CMM` is used to specify the ecological and physiological parameters and the initial conditions for these species as well as the standing stocks of benthos, periphyton, and zooplankton. Each species' daily consumption rate is back-calculated from its expected growth rate using BASS's linear feeding option.

6.2. An Analysis of Everglades Mercury Bioaccumulation

The BASS example project `EX_EVERGLADES_CANAL_HG` simulates methyl mercury contamination in a canal fish community of the Florida Everglades and is constructed as outlined in Section 4.5. For this BASS application largemouth bass (*Micropterus salmoides*), Florida gar (*Lepisosteus platyrhincus*), yellow bullheads (*Ameiurus natalis*), bluegill sunfish (*Lepomis macrochirus*), redear sunfish (*Lepomis microlophus*), and mosquito fish (*Gambusia holbrooki*) are assumed to be the dominant species in the habitats of interest. The sources of the ecological, morphological, and physiological parameters used by this example are documented in its associated FSH files. Turner et al. (1999) reported the mean biomass of large and small fishes across a variety of Everglades habitats to be approximately 60 kg wet wt/ha. Initial standing stocks of the bass, gar, bullheads, bluegill, redear sunfish, and mosquito fish were assigned to be 5, 10, 10, 50, 25, and 5 kg wet wt/ha, respectively, for a total community biomass of 105 kg wet wt/ha. The water concentration of methylmercury for the simulation was assigned to be a constant 0.2 ng/L (Stober et al. 1998) and the BAF's for benthos and zooplankton were assigned to be $10^{6.09}$ and $10^{5.90}$, respectively (Loftus et al. 1998).

At the end of the 10 year simulation, the mean annual standing stocks of the bass, gar, bullheads, bluegill, redear sunfish, and *Gambusia* are 0.867, 1.08, 4.79, 30.0, 35.4, and 2.55 kg wet wt/ha, respectively, for a total community biomass of 74.7 kg wet wt/ha.

The simulated whole-body concentrations of methyl mercury in these species agree reasonably well with unpublished data collected by Lange et al. and Loftus et al. (1998). See **Figure 6.1 - Figure 6.6**. BASS's significant over prediction of the whole-body methylmercury concentrations of redear sunfish is probably due to the specialized feeding behavior of this species. In particular, redear sunfish, which are also known as shellcrackers, often feed almost exclusively on molluscs that generally have significantly lower methylmercury concentration than do other benthic macroinvertebrates. The annual averaged concentrations of methylmercury in

largemouth, gar, bullhead, bluegill, redear and *Gambusia* weighted by cohort biomasses were 0.842, 0.822, 0.580, 0.440, 0.513, and 0.180 mg Hg/kg wet wt, respectively. When weighted by cohort densities, the annual averaged concentrations of methylmercury in largemouth, gar, bullhead, bluegill and redear were 0.450, 0.491, 0.332, 0.237, 0.299, and 0.143 mg Hg/kg wet wt, respectively. Loftus et al. report average whole-body concentrations of methylmercury in largemouth, gar, bullhead, bluegill, redear, and *Gambusia* to be 0.967, 1.16, 0.443-0.755, 0.478, 0.247, and 0.247-0.321 mg Hg/kg wet wt, respectively. The average body weights of largemouth, gar, bullhead, bluegill, redear, and *Gambusia* analyzed by Loftus et al. were 205, 278, 37.5-92.9, 21.8, 73.0, and 0.0602-0.218 g wet wt/fish, respectively.

As is typically observed under field conditions (Forrester et al. 1972, Scott and Armstrong 1972, Cross et al. 1973, Akielaszek and Haines 1981, Watling et al. 1981, Boush and Thieleke 1983a, b, MacCrimmon et al. 1983, Ueda and Takeda 1983, Wren and MacCrimmon 1986, Braune 1987, Luten et al. 1987, Moharram et al. 1987, Sprenger et al. 1988, Grieb et al. 1990, Parks et al. 1991, Gutenmann et al. 1992, Lange et al. 1993, Tracey 1993, Joiris et al. 1995, Munn and Short 1997, Stafford and Haines 1997), **Figure 6.1 - Figure 6.6** predicts a strong interdependence between the body sizes of fish and their whole-body mercury concentrations.

Largemouth bass (*Micropterus salmoides*)

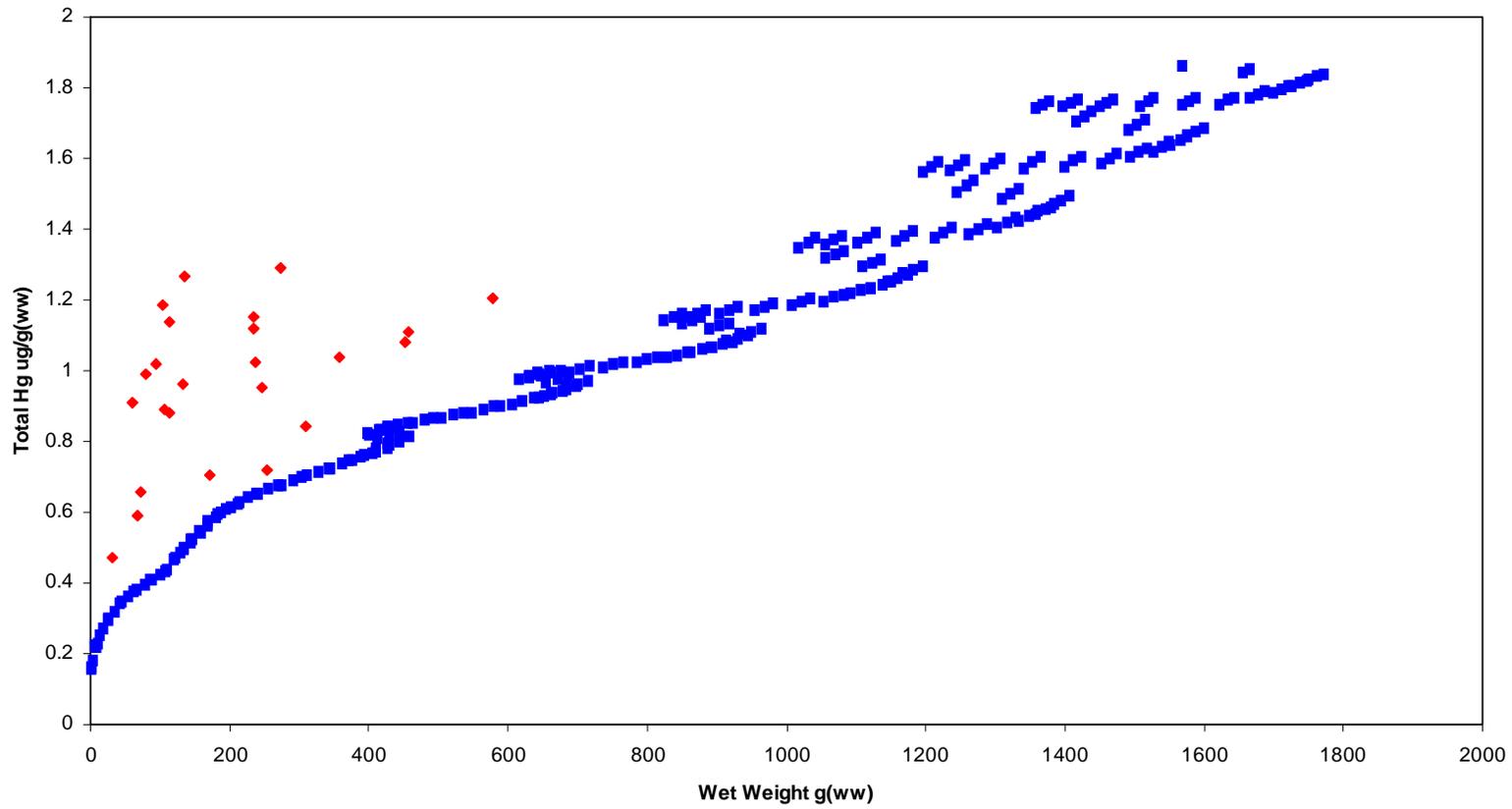


Figure 6.1 Predicted and observed methylmercury concentrations of largemouth bass in Florida Everglades canals.

Florida gar (*Lepisosteus platyrhincus*)

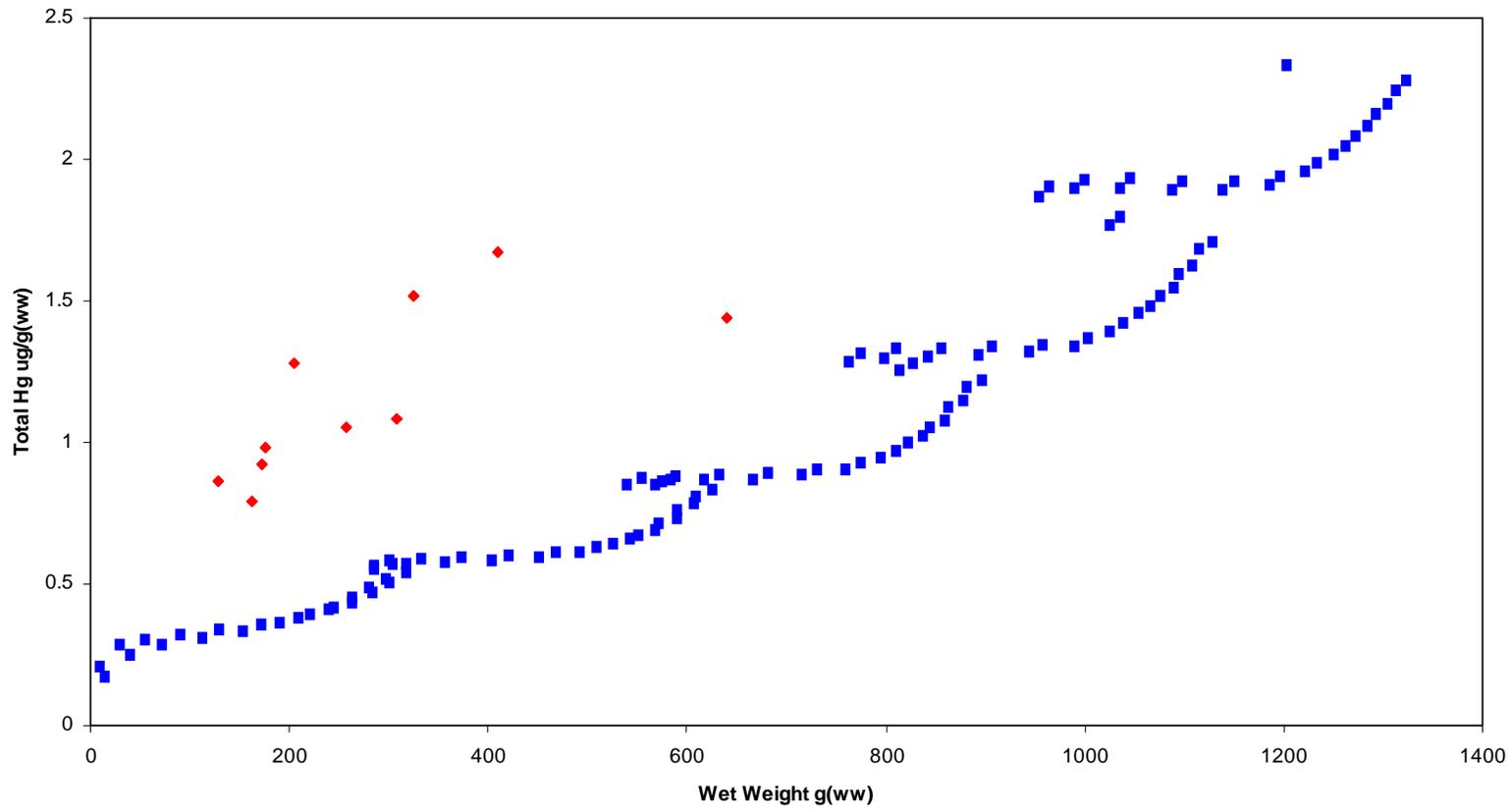


Figure 6.2 Predicted and observed methylmercury concentrations of Florida gar in Florida Everglades canals.

Yellow bullhead (*Ameiurus natalis*)

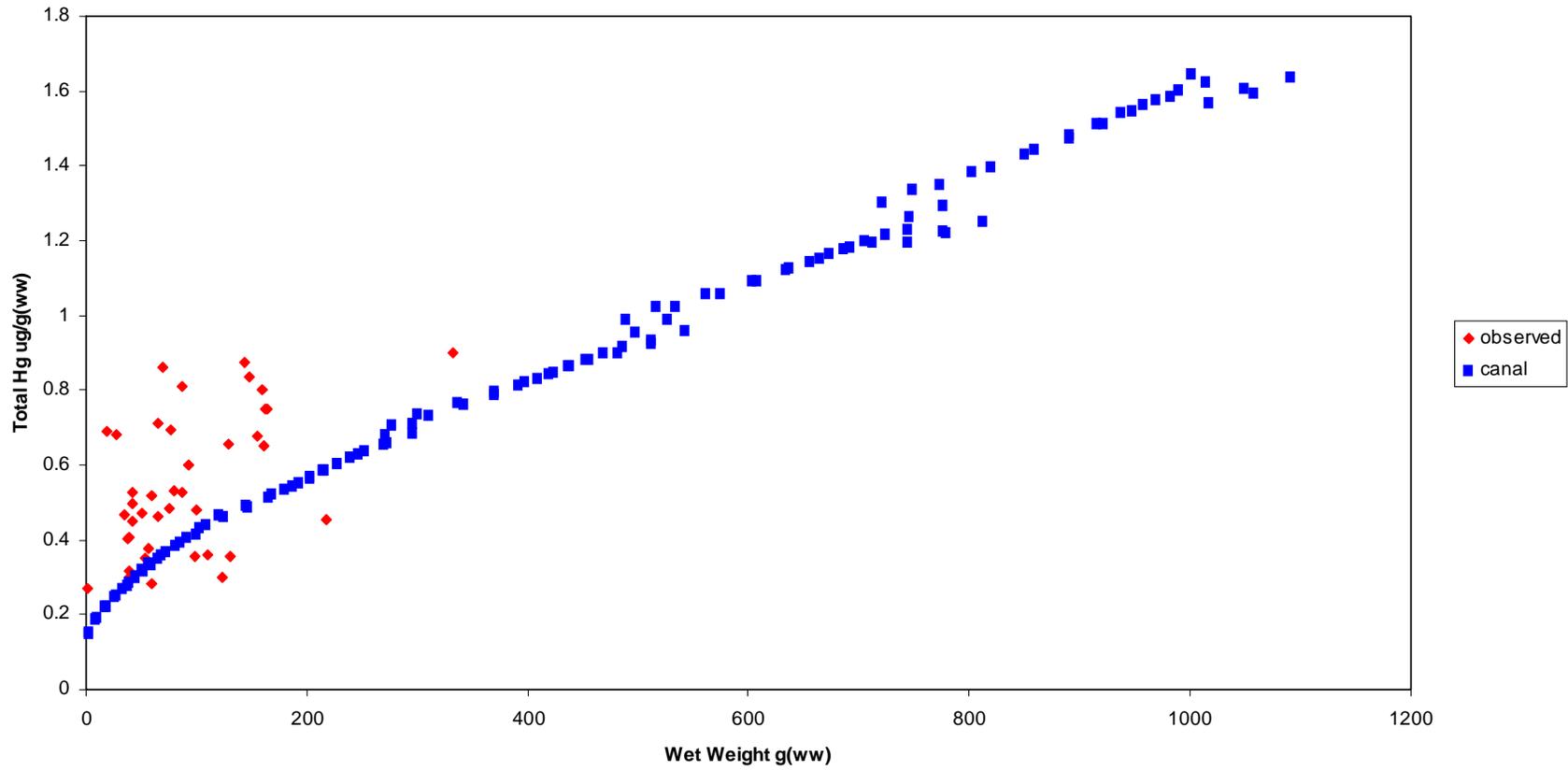


Figure 6.3 Predicted and observed methylmercury concentrations of yellow bullhead in Florida Everglades canals.

Bluegill (*Lepomis macrochirus*)

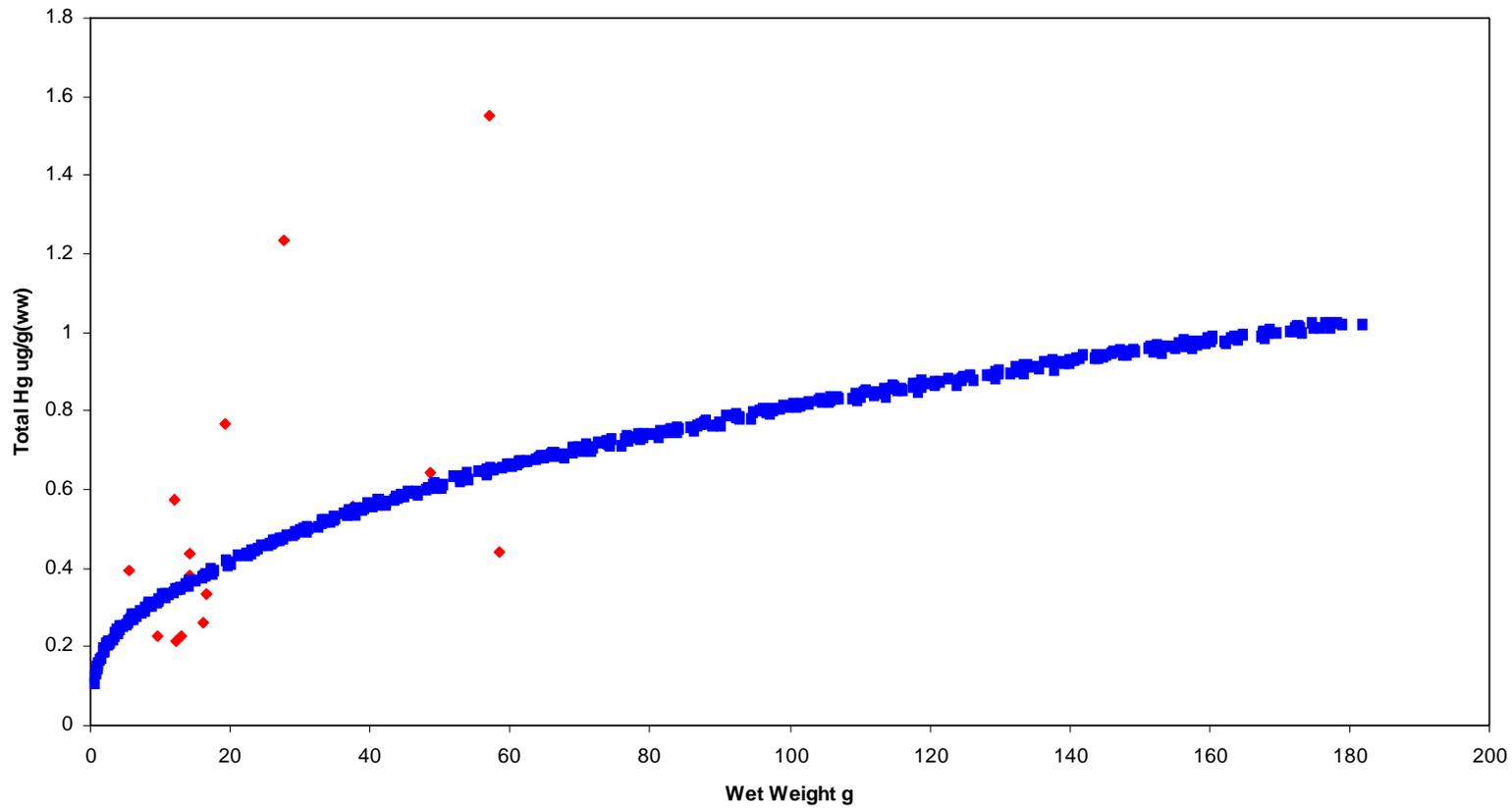


Figure 6.4 Predicted and observed methylmercury concentrations of bluegill sunfish in Florida Everglades canals.

Redear sunfish (*Lepomis microlophus*)

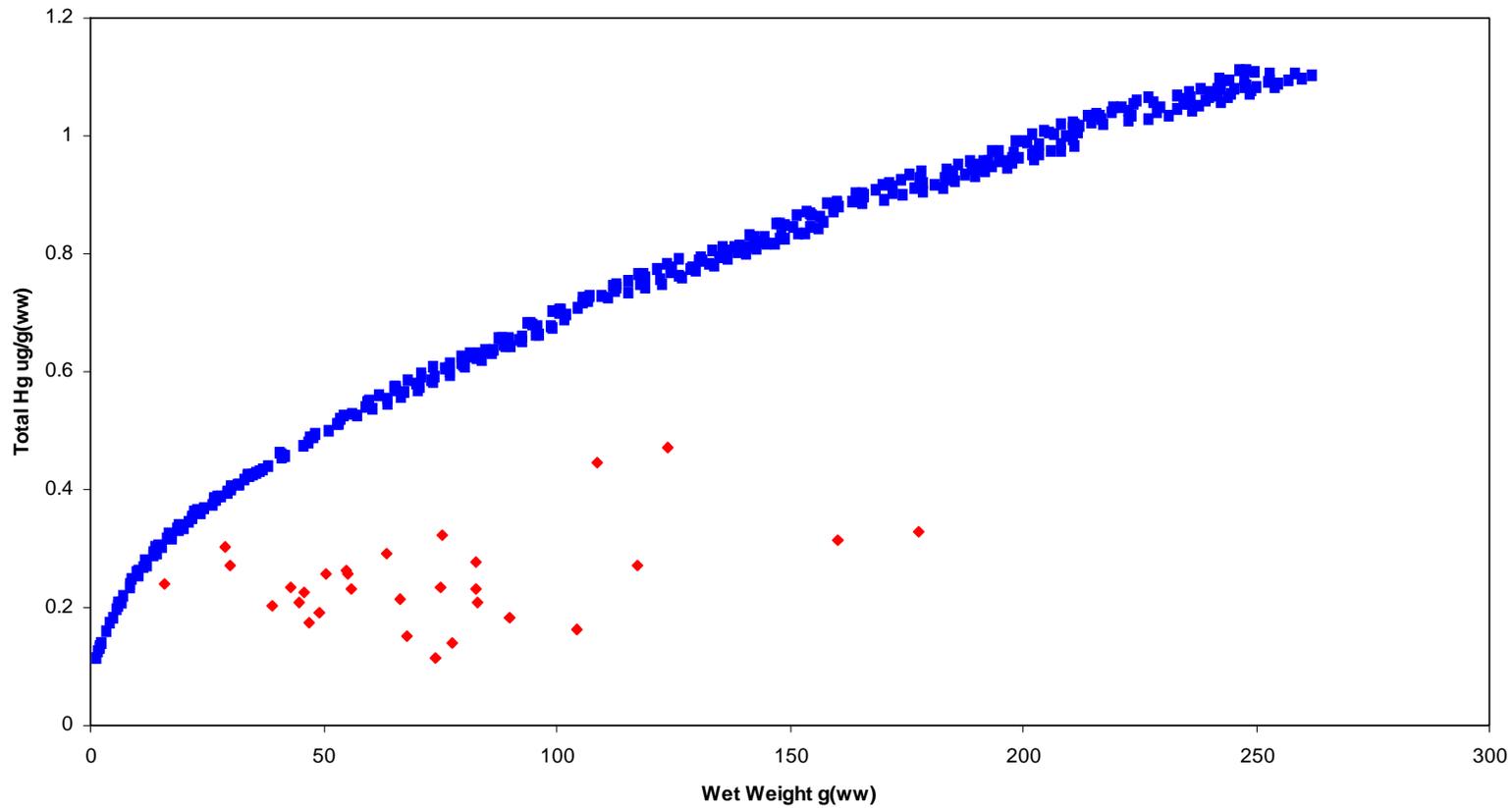


Figure 6.5 Predicted and observed methylmercury concentrations of redear sunfish in Florida Everglades canals.

Mosquitofish (*Gambusia holbrooki*)

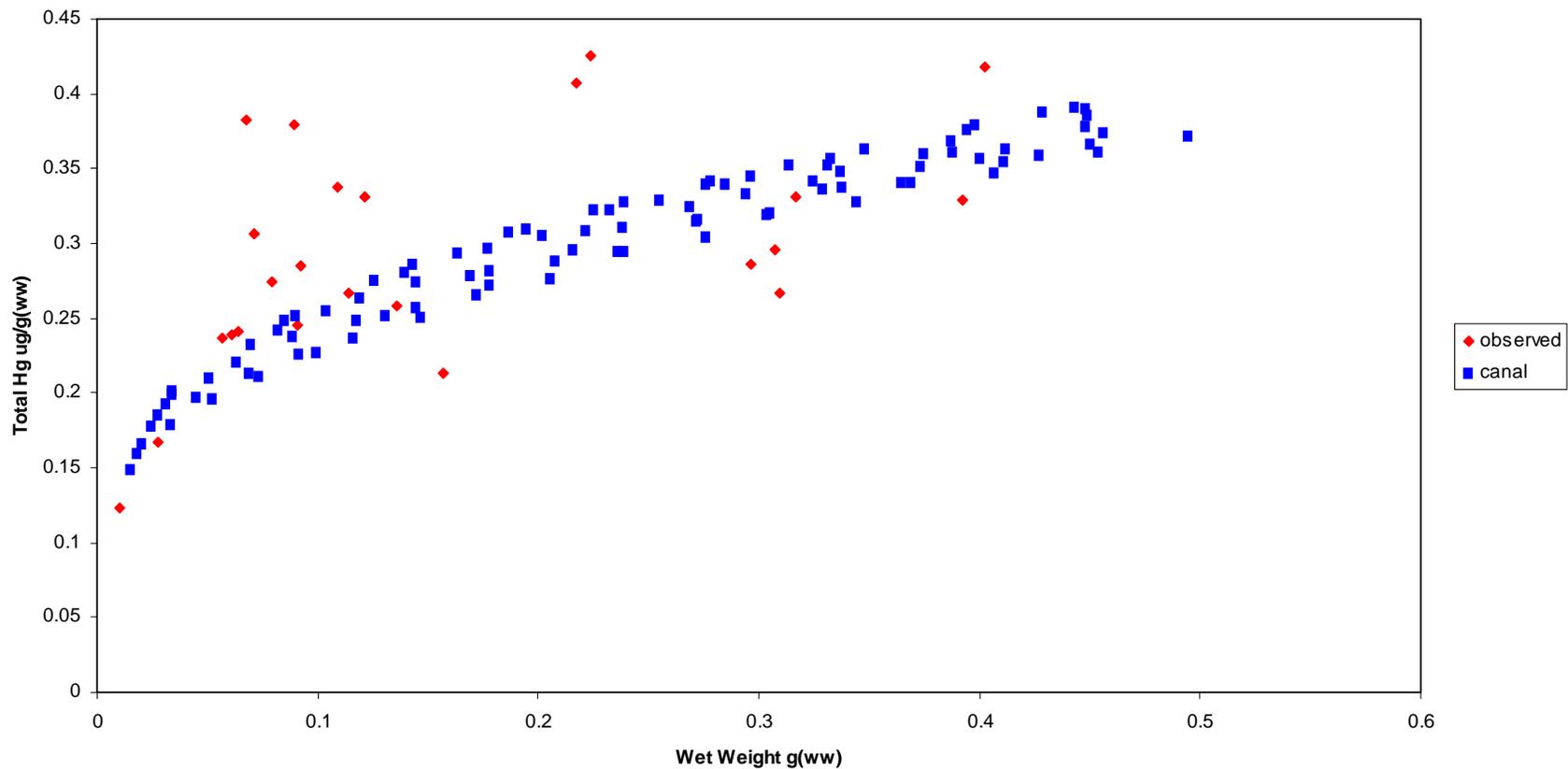


Figure 6.6 Predicted and observed methylmercury concentrations of *Gambusia* in Florida Everglades canals.

7. Model Quality Assurance

Quality Assurance (QA) and Quality Control (QC) for the BASS simulation model has been addressed with respect to:

- 1) The model's theoretical foundations, i.e., does the model's conceptual and mathematical framework stand up to scientific / engineering peer view?
- 2) The model's implementation, i.e., does the code actually do what it is intended to do?
- 3) The model's documentation and application, i.e., can the model be used by the outside research and regulatory community in a meaningful way?

7.1. Questions Regarding QA of a Model's Scientific Foundations

7.1.1. *Is the model's theoretical foundation published in the peer reviewed literature?*

With the exception of its population and trophodynamic algorithms, BASS is based on the FGETS bioaccumulation and bioenergetics model that has been published in the peer reviewed literature (Barber et al. 1988, 1991). These algorithms have been reviewed and compared with other existing bioaccumulation models to document their scientific foundation and to verify their predictive performance (see Barber 2003, 2008). The bioenergetic modeling paradigm that BASS uses to simulate fish growth has been employed by many researchers in the peer reviewed literature (Norstrom et al. 1976, Kitchell et al. 1977, Minton and McLean 1982, Stewart et al. 1983, Thomann and Connolly 1984, Cuenco et al. 1985, Stewart and Binkowski 1986, Beauchamp et al. 1989, Barber et al. 1991, Stewart and Ibarra 1991, Lantry and Stewart 1993, Rand et al. 1993, Roell and Orth 1993, Hartman and Brandt 1995a, Petersen and Ward 1999, Rose et al. 1999, Schaeffer et al. 1999). Since its construction, FGETS has also been included in numerous reviews of bioaccumulation models that are applicable for ecological risk assessments and environmental management (Barron 1990, Jones et al. 1991, Barnthouse 1992, Chapra and Boyer 1992, Landrum et al. 1992, Olem et al. 1992, Dixon and Florian 1993, Wurbs 1994, Cowan et al. 1995, Campfens and Mackay 1997, Feijtel et al. 1997, Deliman and Gerald 1998, Exponent 1998, Howgate 1998, Vorhees et al. 1998, Wania and Mackay 1999, Bartell et al. 2000, Gobas and Morrison 2000, Mackay and Fraser 2000, Bartell 2001, Limno-Tech 2002, Exponent 2003, Sood and Bhagat 2005).

Two criticisms have been lodged against FGETS in the literature. The first of these is that FGETS assumes or attempts to prove that the gill exchange of chemicals is more important than other routes of exchange (Madenjian et al. 1993). Madenjian et al.

(1993) took exception to FGETS predictions that "excretion of PCB through the gills is an important flux in the PCB budget of lake trout". Madenjian et al. claimed that this result was not supported by any laboratory study on trout and cited Weininger (1978) as proof that gill excretion was, in fact, negligible. Nevertheless, Madenjian et al. used a single, unidentified excretion constant in their model that simply lumps all excretion pathways (i.e., gill, intestinal, urinary, and dermal) into one. Thus, what Madenjian et al. are essentially questioning is not FGETS per se but rather the need to use thermodynamically based diffusion models for bioaccumulation in general.

The second criticism is that FGETS is overly complex and requires too much additional data to parameterize (McKim et al. 1994, Stow and Carpenter 1994, Jackson 1996). Since FGETS's bioenergetic model for fish growth is not significantly different from those used by several other authors (Norstrom et al. 1976, Weininger 1978, Thomann and Connolly 1984, Madenjian et al. 1993, Luk and Brockway 1997), this criticism is also generally aimed at BASS's gill exchange model. A recent review and comparison of gill exchange models, however, clearly demonstrated that there is more than ample literature data to parameterize the gill exchange formulations used by FGETS and BASS (Barber 2003).

7.1.2. *How has the model or its algorithms been corroborated or used?*

BASS's dietary and gill exchange algorithms have been corroborated by comparing its predicted dietary assimilation efficiencies and gill uptake and excretion rates to those published in the peer reviewed literature (Barber et al. 1988, Barber 2003, 2008). BASS's dietary exchange algorithms have also been cited by other researchers to explain results of actual exposure studies (e.g., Dabrowska et al. 1996, Doi et al. 2000). For validation of BASS's bioenergetic growth algorithms, the reader is referred to Barber et al. (1991) and the examples herein.

BASS's predictive performance as an integrated bioaccumulation model has been corroborated for organic chemicals by simulations of PCBs dynamics in Lake Ontario salmonids, various laboratory studies, largemouth bass-bluegill-catfish communities of Lake Hartwell / Twelvemile Creek, SC, and Tennessee stream fishes (Barber et al. 1991, USEPA 1994, Brockway et al. 1996, Simon 1999, Marchettini et al. 2001, USEPA 2004). Similarly, Hunt et al. (1992) used FGETS to model DDT bioaccumulation in caged channel catfish at Superfund Sites. For sulfhydryl binding metals, BASS's predictive performance has been corroborated by simulations of methylmercury bioaccumulation in Florida Everglades fish

communities one of which is presented herein as a typical BASS application. Murphy (2004) also successfully used BASS to model and analyze mercury bioaccumulation in the South River and the South Fork of the Shenandoah River in Virginia. More recently, BASS was used to estimate lag times of mercury residues in fish responding to mercury load reductions as part of ORD's review of the Agency's Clean Air Mercury Rule (CAMR, February 15, 2005). This work was subsequently incorporated into the Regulatory Impact Analysis (RIA) that assessed the benefits of atmospheric load reductions to aquatic ecosystems (USEPA 2005).

Several researchers (Lassiter and Hallam 1990, ECOFRAM Aquatic Effects Subcommittee et al. 1998, ECOFRAM 1999, Boxall et al. 2001, Boxall et al. 2002, Reinert et al. 2002) have used BASS's predecessor, FGETS, to predict acute and chronic lethality, and the EPA's Office of Water's AQUATOX modeling system uses the FGETS/BASS lethal effects algorithm as its principal effects module (Park and Clough 2004). Additionally, the Office of Water has recognized BASS as one of the leading models available for simulating time dynamic bioaccumulation for applications when steady-state methods (e.g., BAFs or BSAFs) are considered insufficient (USEPA 2003). The Commonwealth of Virginia has identified BASS as an accepted tool for its PCB bioaccumulation assessments (VDEQ 2005). BASS has also been recommended to the states of Michigan and Washington as an assessment tool (Exponent 1998, 2003).

Whereas Hallam and Deng (2006) implemented the FGETS/BASS bioaccumulation framework within sophisticated McKendrick-von Foerster partial differential equation models for age-structured populations, Cohen and Cooter (2002a, 2002b) incorporated simpler forms of this framework into more holistic fate and transport exposure software. Lastly, Apeti et al. (2005) modified FGETS to simulate metal bioaccumulation in shellfish.

7.1.3. What is the mathematical sensitivity of the model with respect to parameters, state variables (initial value problems), and forcing functions / boundary conditions? What is the model's sensitivity to structural changes?

There are four major classes of mathematical sensitivity regarding a model's behavior. These are the model's sensitivity to parameter changes, forcing functions, initial state variables, and structural configuration. The first three of these classes generally are formally defined in terms of the following partial derivatives

$$\frac{\partial X_i}{\partial p_j}; \quad \frac{\partial X_i}{\partial Z_j}; \quad \frac{\partial X_i}{\partial X_j(0)} \quad (7.1)$$

where X_i is a state variable of interest; p_j is some state parameter of concern; Z_j is some external forcing function; and $X_j(0)$ is the

initial value of some state variable of interest that may be X_i itself. Structural sensitivity, which generally cannot be formulated as a simple partial derivative, typically concerns the number and connectivity between the system's state variables. An excellent question regarding structural sensitivity for a model like BASS might be how does a predator's population numbers or growth rate change with the introduction or removal of new or existing prey items?

Because sensitivity is simply a mathematical characteristic of a model, model sensitivity in and of itself is neither good nor bad. Sensitivity is desirable if the real system being modeled is itself sensitive to the same parameters, forcing functions, initial state perturbations, and structural changes to which the model is sensitive. Even though model sensitivity can contribute to undesirable model uncertainty or prediction error, it is important to acknowledge that model sensitivity and uncertainty are not one and the same (Summers et al. 1993, Wallach and Genard 1998). Model uncertainty, or at least one of its most common manifestations, is the product of both the model's sensitivity to particular components and the statistical variability associated with those components.

A generalized sensitivity analysis of BASS without explicit specification of a fish community of concern is undoable. Furthermore, the results of a sensitivity analysis for one community generally cannot be extrapolated to other communities. Issues related to BASS's sensitivity must be evaluated on a case by case basis by the users of the software. Although procedures for enabling users to conduct a variety of structured sensitivity analyses are currently being developed, presently the onus of performing such analyses rests with the user. Users interested in issues and techniques related to model sensitivity and uncertainty should consult the following papers: Giersch (1991), Elston (1992), Summers et al. (1993), Håkanson (1995), Norton (1996), Loehle (1997), and Wallach and Genard (1998).

7.2. Questions Regarding QA of a Model's Implementation

7.2.1. Did the input algorithms properly process all user input?

As part of its routine output, BASS generates a *.MSG file that summarizes all the input data that were used for a particular simulation. This summary includes not only a line by line summary of the user's input commands but also a complete summary of all control, chemical and fish parameters that BASS assigned based on the user's specified input file(s). The onus is then on the user to verify that their input data has been properly processed. If not, the user should report their problem to the technical contact identified in the BASS user's guide.

BASS has a series of subroutines that check for the completeness and consistency of the user's input data. When missing or inconsistent data are detected, appropriate error messages are written to the *.MSG file, and an error code is set to true. If this error code is true after all the user's input has been processed, BASS terminates without attempting further program execution.

To insure that all program subroutines, functions, and procedures are transmitting and receiving the correct variables, all BASS subroutines and functions are called using implicit interfaces generated by the Lahey / Fujitsu Fortran 95 5.7f compiler. Subroutines and functions are packaged together according to their function and degree of interaction. The BASS v2.2 software is coded with one main program PROGRAM BASS_MAIN (see BASS_PROGRAM.F90) and 33 procedure modules. These modules are:

- MODULE ADAMS_GEAR - subroutines for performing EXAMS Adams-Gear integrations (see EXAMS_ADAM_GEAR.F90).
- MODULE BASS_ALLOC - subroutines for allocating and reallocating derive type pointers (see BASS_ALLOC.F90).
- MODULE BASS_CHECK - subroutines for checking the completeness and consistency of user input (see BASS_CHECK.F90).
- MODULE BASS_DEBUG - subroutines for program debugging. Used only for program development (see BASS_DEBUG.F90).
- MODULE BASS_DEFINED - functions for determining whether program parameters and variables have been initialized or assigned (see BASS_DEFINED.F90).
- MODULE BASS_EXP - subroutines for calculating chemical exposures, community forcing functions, and habitat suitability multipliers (see BASS_EXP.F90).
- MODULE BASS_INI - subroutines for initialization of program variables (see BASS_INI.F90).
- MODULE BASS_INPUT - subroutines for decoding user input (see BASS_INPUT.F90).
- MODULE BASS_INT - subroutines for Adams-Gear, Euler, and Runge-Kutta integrations (see BASS_INT.F90).
- MODULE BASS_INT_LOADER - subroutines for loading BASS derived type variables into standard integration vectors (see BASS_INT_LOADER.F90).
- MODULE BASS_IO - subroutines for processing user input and output (see BASS_IO.F90).
- MODULE BASS_ODE - subroutines for the computational kernel of the BASS software (see BASS_ODE.F90).
- MODULE BASS_PLOTS - subroutines for generating output plots for BASS v2.1 and earlier as well as for code development and maintenance (see BASS_PLOTS.F90).
- MODULE BASS_TABLES - subroutines for generating output tables for BASS v2.1 and earlier as well as for code

- development and maintenance (see BASS_TABLES.F90).
- MODULE BASS_WRITE_CSV - subroutines for generating CSV output files for import into Excel worksheets (see BASS_CSV.F90).
- MODULE BASS_WRITE_XML - subroutines for generating XML output files for post processing by the BASS GUI (see BASS_XML.F90).
- MODULE DECODE_FUNCTIONS - subroutines for decoding constant, linear, and power functions from character strings (see UTL_DCOD_FNC.F90).
- MODULE DISLIN - implicit interfaces for the DISLIN graphics subroutines DISLIN.F90).
- MODULE DISLIN_PLOTS - general utility subroutines for generating 2 and 3-dimensional DISLIN plots (see UTL_PLOTS.F90).
- MODULE ERROR_MODULE - subroutines for printing error codes encountered with general utility modules (see UTL_ERRORS.F90).
- MODULE FILESTUFF - subroutines for parsing file names and obtaining version numbers or time stamps (see UTL_FILESTUFF.F90).
- MODULE FLOATING_POINT_COMPARISONS - operators for testing equality or inequality of variables with explicit consideration of their computer representation and spacing characteristics (see UTL_FLOATCMP.F90).
- MODULE GETNUMBERS - subroutines for extracting numbers from character strings (see UTL_GETNUMS.F90).
- MODULE IOSUBS - subroutines for assigning, opening, and closing logical units (see UTL_IOSUBS.F90).
- MODULE MODULO_XFREAD - subroutines for reading files that contain comments, continuation lines, and include files (see UTL_XFREAD.F90).
- MODULE MSORT - subroutines for sorting and generating permutation vectors for lists and vectors (see UTL_MSORT.F90).
- MODULE MXGETARGS - subroutines for extracting arguments from a command line (see UTL_MXGETARGS.F90).
- MODULE REALLOCATER - subroutines for allocating and reallocating integer, logical, and real pointers (see UTL_ALLOC.F90).
- MODULE SEARCH - subroutines for finding the location of a key phrase within a sorted list (see UTL_SEARCH.F90).
- MODULE SEARCH_LISTS - subroutines for finding the location of a value within a sorted list (see UTL_SEARCH_LISTS.F90).
- MODULE STRINGS - subroutines for character string manipulations and printing multiline character text (see UTL_STRINGS.F90).
- MODULE TABLE_UTILS - subroutines for generating self-formatting tables (see UTL_PTABLE.F90).
- MODULE UNITSLIBRARY - subroutines for defining and

performing units conversions (see UTL_UNITSLIB.F90).

In general, these procedure modules are coded with minimal scoping units. Consequently, their component subroutines and functions explicitly initialize all required internal variables. This safeguard is intended to prevent inadvertent use of uninitialized variables. Whenever possible, subroutine and function arguments are declared with INTENT(IN) and INTENT(OUT) declarations to preclude unintentional reassignments.

Although global constants and Fortran parameters are supplied to program procedures via modules (see question 7.2.3), data exchanges between program procedures are performed via formal subroutine / function parameters whenever possible. The only notable exceptions to this coding policy are modules that must be used to supply auxiliary parameters to “external” subroutines that are used as arguments to certain mathematical subroutines (e.g., root finding subroutines). Working areas used by BASS are not used for data transfers between internal and external procedures.

To simplify the construction and maintenance of the formal parameter lists of many BASS subroutines and functions and to help prevent the inadvertent transposition of formal parameters, BASS makes extensive use of derived type data structures. Each derived type definition is specified within its own module, and all derived type definition modules are maintained in a single file (BASS_TYPES.F90.) Derived types used by BASS v2.2 are:

- MODULE BASS_TYPE_CHEM_PAR - type definition for chemical parameters
- MODULE BASS_TYPE_DIET_MEAN - type definition used to summarize average realized diets.
- MODULE BASS_TYPE_DIET_PAR - type definition used by derived type BASS_TYPE_FOODWEB_PAR
- MODULE BASS_TYPE_DIETS - type definition used for input processing of user-specified fish diets
- MODULE BASS_TYPE_FISH_INT - type definition for integrated fish variables and fluxes
- MODULE BASS_TYPE_FISH_PAR - type definition for fish parameters
- MODULE BASS_TYPE_FISH_VAR - type definition for current fish variables and fluxes
- MODULE BASS_TYPE_FOODWEB_PAR - type definition for the decoded user-supplied fish diets and community trophic structure.
- MODULE BASS_TYPE_HSI_PAR - type definition for fish habitat multipliers
- MODULE BASS_TYPE_NONFISH_INT - type definition for integrated nonfish variables and fluxes
- MODULE BASS_TYPE_NONFISH_PAR - type definition for nonfish parameters
- MODULE BASS_TYPE_NONFISH_VAR - type definition for

current nonfish variables and fluxes

- MODULE BASS_TYPE_PLOT_DATA - type definition for user-specified plots
- MODULE BASS_TYPE_PREY_ITEMS - type definition used by derived type BASS_TYPE_FISH_VAR to store a fish’s currently realized dietary composition
- MODULE BASS_TYPE_QSAR_DATA - type definition for linked list used during data input
- MODULE BASS_TYPE_QSAR_LINKED_LIST - type definition for linked list used during data input
- MODULE BASS_TYPE_QSAR_NODE - type definition for linked list used during data input
- MODULE BASS_TYPE_TROPHIC - definition used for the calculation of realized diet composition and consumption
- MODULE BASS_TYPE_VMATRIX_LOGICAL - type definition for DISLIN graphing matrices
- MODULE BASS_TYPE_VMATRIX_REAL - type definition for DISLIN graphing matrices
- MODULE BASS_TYPE_ZFUNCTION_PAR - type definition for user-supplied exposure and forcing functions

A good example of BASS’s use of derived type data structures is the derived type variable used to store and transfer the ecological, physiological, and morphometric data for a particular fish species. This derived type is defined by following module

```
MODULE bass_type_fish_par
USE bass_type_hsi_par
TYPE:: fish_par
  CHARACTER (LEN=80) :: ageclass, ast_type, ast_var, common_name, &
    fmodel_var, genus_species, spawning_interval, temp_var
  INTEGER :: fmodel_cls=0, harvests=0, spawnings=0, &
    stockings=0, temperatures=0
  INTEGER, DIMENSION(:), POINTER :: fmodel=>NULL()
  INTEGER, DIMENSION(:), POINTER :: spawn_dates=>NULL()
  INTEGER, DIMENSION(:), POINTER :: harvest_date1=>NULL()
  INTEGER, DIMENSION(:), POINTER :: harvest_date2=>NULL()
  INTEGER, DIMENSION(:), POINTER :: stock_dates=>NULL()
  LOGICAL :: bb_constant=.TRUE., prey_switching_on=.TRUE.
  REAL :: ae_fish, ae_invert, ae_plant, ast_bb, ast_bnds, ast_pop, &
    dry2live_ab, dry2live_aa, dry2live_bb, dry2live_cc, gco2_d, kf_min, &
    la, longevity, mgo2_s, rbi, refugia, rq, rt2std, sda2in, tl_r0, wt_max, yoy
  REAL, DIMENSION(2) :: ga, id, ld, ll, lw, pa, pl, sg_mu, wl
  REAL, DIMENSION(3) :: nm
  REAL, DIMENSION(4) :: lp, lp_max, lp_min
  REAL, DIMENSION(5) :: ge, mf, mi, sg, sm, so, st
  REAL, DIMENSION(:), POINTER :: fmodel_bnds=>NULL()
  REAL, DIMENSION(:), POINTER :: harvest_len1=>NULL()
  REAL, DIMENSION(:), POINTER :: harvest_len2=>NULL()
  REAL, DIMENSION(:), POINTER :: harvest_rate=>NULL()
  REAL, DIMENSION(:), POINTER :: stock_age=>NULL()
  REAL, DIMENSION(:), POINTER :: stock_rate=>NULL()
  REAL, DIMENSION(:), POINTER :: stock_tl=>NULL()
  REAL, DIMENSION(:), POINTER :: stock_wt=>NULL()
  REAL, DIMENSION(:), POINTER :: temp_bnds=>NULL()
  REAL, DIMENSION(:), POINTER :: temp_pref=>NULL()
  TYPE(hsi_par) :: hsi_feed, hsi_persist, hsi_recruit
END TYPE fish_par
END MODULE bass_type_fish_par
```

Many components of this derived type are user input parameters that have already been discussed. For example, the array `ga(2)` stores the coefficient and exponent of a species' gill area function (see /MORPHOMETRIC_PARAMETERS page 49). Other components are secondary parameters that are calculated from the user's input data. For example, `dry2live_ab`, `dry2live_aa`, `dry2live_bb`, and `dry2live_cc` are constants that are used to calculate a fish's live weight from its dry weight (see introduction to Section 2.6. Modeling Growth of Fish). Using a declaration of the form

```
TYPE(fish_par), DIMENSION(nspecies) :: par
```

all data defined by the above derived type can be passed to a BASS subroutine by the simple calling statement

```
CALL sub1(...., par, ....)
```

without fear of data misalignment.

To insure that all program subroutines, functions, and procedures use the same global constants or parameters, such constants are declared and defined within a set of 15 data modules. These modules include:

- MODULE ADAM_DATA - stores control parameters for the EXAMS Adams-Gear integrators (see EXAMS_ADAM_GEAR_MODULES.F90).
- MODULE BASS_CONSTANTS - specifies various biological and physical constants used by BASS's computational subroutines (see BASS_GLOBALS.F90).
- MODULE BASS_GRAETZ - specifies parameters used to calculate chemical exchange across the fish gills (see BASS_GLOBALS.F90).
- MODULE BASS_IOFILES - specifies logical unit numbers for input and output devices (see BASS_GLOBALS.F90).
- MODULE BASS_NAMES - stores user-supplied fish and chemical names (see BASS_GLOBALS.F90).
- MODULE BASS_NOVALUE - specifies values for integer, real, and character variables that have not been initialized (see BASS_GLOBALS.F90).
- MODULE BASS_PRECISION - specifies the precision of floating point variables as either single, double, or quad precision variables. This module also assigns certain associated floating point constants (see BASS_GLOBALS.F90).
- MODULE BASS_UNITS - specifies unit conversion factors that are specific to BASS for use by MODULE UNITSLIBRARY (see BASS_UNITS.F90).
- MODULE BASS_WORKING_DIMENSIONS - specifies "standard" sizes for character variables, input records, etc. (see BASS_GLOBALS.F90).
- MODULE CONSTANTS - constants used by utility

- subroutines (see UTL_CONSTANTS.F90).
- MODULE GEAR_DATA - stores control parameters for the EXAMS Adams-Gear integrators (see EXAMS_ADAM_GEAR_MODULES.F90).
- MODULE LOCAL_GEAR_DATA - stores control parameters for the EXAMS Adams-Gear integrators (see EXAMS_ADAM_GEAR_MODULES.F90).
- MODULE STEP_DATA - stores control parameters for the EXAMS Adams-Gear integrators (see EXAMS_ADAM_GEAR_MODULES.F90).
- MODULE STIFF_DATA - stores control parameters for the EXAMS Adams-Gear integrators (see EXAMS_ADAM_GEAR_MODULES.F90).
- MODULE UNITS_PARAMETERS - specifies parameters used by the units conversion subroutines (see UTL_UPARAMS.F90)

BASS v2.2 uses the following modules (see BASS_WORK_AREAS.F90) to define work areas that are common to two or more functions or subroutines.

- MODULE BASS_CPU_PERFORMANCE
- MODULE BASS_FOODWEB_WORK_AREA
- MODULE BASS_HSI_MEANS
- MODULE BASS_MULTISORT_WORK_AREA
- MODULE BASS_ODE_WORK_AREA
- MODULE BASS_OUTPUT_WORK_AREA
- MODULE BASS_PLOT_WORK_AREA

7.2.3. Is the developer reasonably confident that all program subroutines, functions, and procedures are using the same global constants or parameters?

All global constants are defined within their own individual modules. These modules include

- MODULE BASS_CONSTANTS - constants used by BASS's computational subroutines (see BASS_GLOBALS.F90).
- MODULE BASS_NOVALUE - specifies values for integer, real, and character variables that have not been initialized (see BASS_GLOBALS.F90).
- MODULE BASS_PRECISION - specifies the precision of floating point variables as either single, double, or quad precision variables. This module also assigns certain associated floating point constants (see BASS_GLOBALS.F90).
- MODULE BASS_WORKING_DIMENSIONS - specifies "standard" sizes for character variables, input records, etc. (see BASS_GLOBALS.F90).
- MODULE CONSTANTS - constants used by utility subroutines (see UTL_CONSTANTS.F90).
- MODULE UNITS_PARAMETERS - specifies parameters used

by the units conversion subroutines (see UTL_UPARAMS.F90)

7.2.4. Do all strictly mathematical algorithms do what they are supposed to? For example, are root finding algorithms functioning properly?

During execution, BASS must employ root finding algorithms for two important types of calculations. The first of these is the calculation of a fish's live weight from its dry weight given an allometric relationship between its live body weight and its fraction lipid, and linear relationships between its moisture, lipid, and non-lipid organic matter fractions. The second type of calculation involves the linear transformation of unconditioned dietary electivities into self-consistent sets of dietary electivities. These calculations are performed using the combined bisection / Newton-Raphson algorithm outlined by Press et al. (1992).

As mentioned earlier, BASS's differential equations are integrated using a fifth-order Runge-Kutta method with adaptive step sizing that monitors the accuracy of its integration. BASS's Runge-Kutta integrator is patterned on the fifth-order Cash-Karp Runge-Kutta algorithm outlined by Press et. al. (1992) and was tested using the following system of equations.

$$\begin{aligned}
 \frac{dy_1}{dx} &= 1.0 \\
 \frac{dy_2}{dx} &= x \\
 \frac{dy_3}{dx} &= \cos(x) \\
 \frac{dy_4}{dx} &= \cosh(x) \\
 \frac{dy_5}{dx} &= \exp(x) \\
 \frac{dy_6}{dx} &= 1.0/(1.0 + x) \\
 \frac{dy_7}{dx} &= 1.0/(1.0 + x^2) \\
 \frac{dy_8}{dx} &= 1.0/\sqrt{1.0 + x^2} \\
 \frac{dy_9}{dx} &= -100(y_9 - \sin(x)) & y_9(0) &= 1 \\
 \frac{du}{dx} &= 998u + 1998v & u(0) &= 1 \\
 \frac{dv}{dx} &= -999u - 1999v & v(0) &= 0
 \end{aligned}
 \tag{7.2}$$

The analytical solution to this system of equations is

$$\begin{aligned}
 y_1 &= x - x_0 \\
 y_2 &= 0.5(x^2 - x_0^2) \\
 y_3 &= \sin(x) - \sin(x_0) \\
 y_4 &= \sinh(x) - \sinh(x_0) \\
 y_5 &= \exp(x) - \exp(x_0) \\
 y_6 &= \ln(1 + x) - \ln(1 + x_0)
 \end{aligned}
 \tag{7.3}$$

$$\begin{aligned}
 y_7 &= \arctan(x) - \arctan(x_0) \\
 y_8 &= \operatorname{asinh}(x) - \operatorname{asinh}(x_0) \\
 y_9 &= \frac{10101}{10001} \exp(-100x) \\
 &\quad - \frac{100}{10001} \cos(x) + \frac{10000}{10001} \sin(x) \\
 u &= 2 \exp(-x) - \exp(-1000x) \\
 v &= -\exp(-x) + \exp(-1000x)
 \end{aligned}$$

On the interval $[0 < x < 10]$, the above solutions range in value from $v=0.453999E-04$ to $y_5=0.220255E+05$. Besides their large numerical range, the last three equations in this system are numerically stiff (Press et al. 1992, Ascher and Petzold 1998). When integrated on the interval $[0 < x < 10]$, the ratio of the numerical solutions and the corresponding analytical solutions equaled unity with an absolute error of $< 10^{-6}$.

BASS's Runge-Kutta algorithm has also been compared to the adaptive Adams-Gear algorithm employed by the widely used EXAMS fate and transport model. These comparisons demonstrated that BASS's Runge-Kutta algorithm was not only as accurate as the EXAMS Adams-Gear algorithm but was also computationally faster.

7.2.5. Are mathematical algorithms implemented correctly, i.e., are the assumptions of the procedure satisfied by the problem of interest?

Because BASS is a differential equation model, a question of paramount concern is how its integration between points of discontinuity / nondifferentiability is controlled. BASS, like many ecological models, utilizes threshold responses, absolute value functions, maximum and minimum functions, and linear interpolations between time series in its formulation and implementation. Although most of BASS's parameters are updated continuously, some parameters that change very slowly and that are computationally intensive to evaluate (e.g., dietary compositions) are updated only daily. All of these features create points of discontinuity or nondifferentiability. Although there is nothing intrinsically wrong with using such formulations in differential equation models, numerical integrations of such models must proceed from one point of discontinuity / nondifferentiability to another.

With these considerations in mind, BASS's computational kernels (subroutines BASS_ODESOLVR and FGETS_ODESOLVR) are designed to integrate BASS's differential equations for a single day of the desired simulation period. Immediately following the call of these computational kernels, BASS calculates the dietary composition of each fish that will be held constant for that day. The progress of the subsequent numerical integration within the

day is then controlled by any condition that results in a point of nondifferentiability. The two most important conditions in this regard occur when BASS must read an exposure file to update the parameters for the linear interpolation of one or more exposure variables, or when one or more cohorts are eliminated from the community. In the later case, BASS also recalculates the dietary compositions of the remaining fish that again will remain constant for the remainder of the day. Note that recruitment of new cohorts into the simulated community does not create a point of nondifferentiability for BASS since such amendments to the community's structure are performed before calling the computational kernels BASS_ODESOLVR or FGETS_ODESOLVR and, therefore, constitutes a simple reinitialization problem.

7.2.6. Are simulated results consistent with known mathematical constraint of the model? For example, if state variables are supposed to be non-negative, are they? Similarly, if the model is supposed to mass balance, does it?

BASS's state variables, like those of most physical or biological models, must be by definition non-negative. However, insuring that the numerical integration of a differential equation model remains constrained to its appropriate state space is not a trivial issue. Consider, for example, the case when one wants to take a simple Eulerian step for a non-negative state variable that has a negative derivative. If the state variable is to remain non-negative, then the largest allowable size for the integration step can be calculated as follows

$$\begin{aligned}
 y(t+h) &= y(t) + h y'(t) \\
 0 &< y(t) + h y'(t) \\
 \frac{-y(t)}{y'(t)} &> h \quad \text{where } y'(t) < 0
 \end{aligned}
 \tag{7.4}$$

If h is greater than the numerical spacing of t (i.e., $t + h \neq t$), then an integration step is possible. If the converse is true, however, the function $y(t)$ is approximating a step function in which case the desired integration can simply be restarted with $y(t) = 0$. There are at least two types of situations that can occur during a BASS simulation that might necessitate this type of corrective action. The first of these occurs when a cohort experiences intense predation or other mortality that drives its population to extinction whereas the second situation might occur when there is the rapid excretion of a hydrophilic contaminant following the disappearance of an aqueous exposure. When the derivative for a fish's body weight, population density, or body burden is negative, BASS verifies whether the current integration step will, in fact, yield non-negative state values. If not, BASS either executes a simple Euler step of the appropriate size or restarts the integration with the appropriate state variables initialized to zero.

Using the "-mba" command line option, BASS performs a comprehensive mass balance analysis of its fundamental differential equations (i.e., Equations (2.1), (2.2), and (2.3)). BASS also calculates and reports mass balances for each cohort's total biomass and the community's total predicted predatory mortality and its total predicted piscivorous consumption. For the example presented herein, this mass balance is $-2.950E-09$ g dry wt/ha/yr. Since this community's total piscivory is calculated to be $8.850E+03$ g dry wt/ha/yr, this mass balance check would have a relative error of less than 10^{-11} .

7.2.7. Are simulation results consistent across machines or compilers?

BASS was originally developed on a DEC 3000 work station using the DEC Fortran 90 compiler. In November 1999, it was ported to the Windows operating system on the DELL OptiPlex using the Lahey / Fujitsu Fortran 95 5.7f compiler. Although the results of these two implementations agree with one another up to single precision accuracy, due to differences in compiler optimization, model computations must be performed in double precision to obtain this level of consistency.

In September 2004, BASS was ported to a IBM Intellistation A Pro workstation equipped with dual 64-byte Opteron processors and a Windows XP operating system. The BASS source code was then recompiled using the Absoft multiprocessor Fortran 90/95 compilers 8.2 MP and 9.0 MP. Although initial compilations using these compilers failed due to compiler bugs that have been acknowledged by Absoft Technical Support, workarounds for these bugs were successfully implemented. Simulation results of the BASS Absoft MP dual processor executables were in excellent agreement with those of the BASS Lahey-Fujitsu single processor executable. With respect to execution times:

- 1) BASS Lahey-Fujitsu executable runs on standard EPA, single processor machines were approximately 1.5 times slower than BASS Lahey-Fujitsu executable runs on the dual processor workstation.
- 2) BASS Lahey-Fujitsu executable runs on standard EPA, single processor machines were approximately 2.9 times slower than BASS Absoft MP executable runs on standard EPA, single processor machines
- 3) BASS Lahey-Fujitsu executable runs on standard EPA, single processor machines were approximately 5.2 times slower than BASS Absoft MP SOF executable runs on the dual processor workstation.

7.2.8. Have test and reference / benchmark data sets been documented and archived?

The ten BASS projects discussed in Section 6.1 serve not only as BASS distribution examples but also as test projects that track changes in the operation of BASS associated with code maintenance and updates. These project files are used as benchmarks to verify that code modifications that should not change BASS's computational results do not change BASS's simulation output.

7.3. Questions Regarding QA of Model Documentation and Applications

7.3.1. Is the model intended for absolute or comparative prediction?

Although BASS can be used to analyze results from actual field studies or predict the expected future condition of specific real communities, its principal intended use is to predict and compare the outcomes of alternative management options that are associated with pollution control, fisheries management, and / or ecosystem restoration activities.

7.3.2. Does the User Guide provide the information needed to appropriately apply and use the model?

The BASS User's Guide summarizes the model's theoretical foundations and assumptions, the model's input command structure, issues related to user file and project management, and software installation. The User's Guide also presents and discusses the results of one of eight example applications that are distributed with the BASS software.

7.3.3. What internal checking can be made to help insure that the model is being used appropriately?

Currently, the only internal checking performed by BASS is to verify that all parameters needed by the model for a particular simulation have, in fact, been specified by the user. Although BASS does assign default values for a limited number of parameters, most unassigned parameters are fatal errors. Future versions of BASS will perform bounds checking on many of its physiological and morphological parameters.

7.3.4. Has the developer anticipated computational problem areas that will cause the model to "bomb"?

Several key mathematical calculations have been identified as potential problem areas for a BASS simulation. In general, these problem areas involve either the unsuccessful resolution of a root of a nonlinear equation or the unsuccessful integration of BASS's basic state variables. Examples of the former include situations when BASS's calculated dietary compositions do not sum to unity or when a fish's live weight is calculated to be less or equal to its dry weights. Examples of the latter include situations when the current integration step is less than the numerical spacing of the current time point, or when BASS's integration error exceeds 10^{-5} . When these situations are encountered, BASS terminates execution and issues an appropriate error message to the current *.MSG file.

8. Planned Future Features

Presently, ten major program developments are planned for BASS. These include:

- Development of canonical fish and community databases (i.e., *.FSH and *.CMM files) to facilitate easier application of BASS.
- Software to perform model sensitivity analyses.
- Implementation of an option to read a simulated or measured time series of dissolved oxygen concentrations that are needed to calculate the fishes' ventilation volumes. See Equation (2.12). Currently, BASS uses saturated dissolved oxygen concentrations that are calculated as a function of water temperature.
- Development of submodels for simulating the physiological tolerances of fish to water quality parameters other than toxic chemicals.
- Incorporation of quantitative structure activity relationships (QSAR's) to predict metabolism of organic chemicals.
- Development of immigration algorithms for simulating the movement of fish into the simulated community based on habitat parameters such as water depth, current velocity, availability of prey, etc.
- Development of subroutines to simulate sublethal, residue-based effects.
- Enable lipid fractions, fecundity, and physiological mortality to be functionally dependent on the fish's predicted growth rate and/or duration of fasting (see Adams and Huntingford 1997, Simpkins et al. 2003).
- Enable an option for specifying habitat suitability multipliers on respiratory expenditures. See for example Sweka and Hartman (2001) and Facey and Grossman (1990).
- Implementation of light and nutrient dependent primary production by phytoplankton and periphyton.

REFERENCES

- Abernethy, S.G., D. Mackay, and L.S. McCarty. 1988. "Volume fraction" correlation for narcosis in aquatic organisms: the key role of partitioning. *Environmental Toxicology and Chemistry* 7: 469-481.
- Adams, C.E. and F.A. Huntingford. 1997. Growth, maturation and reproductive investment in Arctic charr. *Journal of Fish Biology* 51: 750-759.
- Adey, G., B. Wardley-Smith, and D. White. 1976. Mechanism of inhibition of bacterial luciferase by anaesthetics. *Life Sciences* 17: 1849-1854.
- Akielaszek, J.J. and T.A. Haines. 1981. Mercury in the muscle tissue of fish from three northern Maine lakes. *Bulletin of Environmental Contamination and Toxicology* 27: 201-208.
- Allen, M.S., M.V. Hoyer, and D.E. Canfield, Jr. 1998. Factors related to black crappie occurrence, density, and growth in Florida Lakes. *North American Journal of Fisheries Management* 18: 864-871.
- Apeti, D.A., E. Johnson, and L. Robinson. 2005. A model for bioaccumulation of metals in *Crassostrea virginica* from Apalachicola Bay, Florida. *American Journal of Environmental Sciences* 1: 239-248.
- Arnot, J.A. and F.A.P.C. Gobas. 2004. A food web bioaccumulation model for organic chemicals in aquatic ecosystems. *Environmental Toxicology and Chemistry* 23: 2343-2355.
- Ascher, U.M. and L.R. Petzold. 1998. *Computer Methods for Ordinary Differential Equations and Differential-Algebraic Equations*. SIAM, Philadelphia.
- Atkins, P.W. 1978. *Physical Chemistry*. W.H. Freeman and Company, San Francisco.
- Bailey, H.C., J.L. Miller, M.J. Miller, L.C. Wiborg, L. Deanovic, and T. Shed. 1997. Joint acute toxicity of diazinon and chlorpyrifos to *Ceriodaphnia dubia*. *Environmental Toxicology and Chemistry* 16: 2304-2308.
- Barber, M.C. 2003. A review and comparison of models for predicting dynamic chemical bioconcentration in fish. *Environmental Toxicology and Chemistry* 22: 1963-1992.
- Barber, M.C. 2008. Dietary uptake models used for modeling the bioaccumulation of organic contaminants in fish. *Environmental Toxicology and Chemistry* 27: 755-777.
- Barber, M.C., L.A. Suárez, and R.R. Lassiter. 1987. "FGETS" (Food and Gill Exchange of Toxic Substances): A simulation model for predicting the bioaccumulation of nonpolar organic pollutants by fish. U.S. Environmental Protection Agency, Office of Research and Development, Athens, GA. EPA/600/3-87/038.
- Barber, M.C., L.A. Suárez, and R.R. Lassiter. 1988. Modeling bioconcentration of nonpolar organic pollutants by fish. *Environmental Toxicology and Chemistry* 7: 545-558.
- Barber, M.C., L.A. Suárez, and R.R. Lassiter. 1991. Modelling bioaccumulation of organic pollutants in fish with an application to PCBs in Lake Ontario salmonids. *Canadian Journal of Fisheries and Aquatic Sciences* 48: 318-337.
- Barnthouse, L.W. 1992. The role of models in ecological risk assessment: a 1990's perspective. *Environmental Toxicology and Chemistry* 11: 1751-1760.
- Barrett, J.C. 1992. Turbidity-induced changes in reactive distance of rainbow trout. *Transactions of the American Fisheries Society* 121: 437-443.
- Barron, M.G. 1990. Bioconcentration. *Environmental Science and Technology* 24: 1612-1618.
- Bartell, S.M. 2001. Chapter 9. Ecosystem Models - Aquatic. *In Ecological Modeling in Risk Assessment: Chemical Effects on Populations, Ecosystems, and Landscapes. Edited by R.A. Pastorok, S.M. Bartell, S. Ferson and L.R. Ginzburg*. CRC Press, Lewis Publishers, Boca Raton, FL.
- Bartell, S.M., K.R. Campbell, C.M. Lovelock, S.K. Nair, and J.L. Shaw. 2000. Characterizing aquatic ecological risks from pesticides using a diquat dibromide case study III: ecological process models. *Environmental Toxicology and Chemistry* 19: 1441-1453.
- Beamish, F.W.H. and M. Legrow. 1983. Bioenergetics of the southern brook lamprey, *Ichthyomyzon gagei*. *Journal of Animal Ecology* 52: 575-590.
- Beamish, F.W.H. and P.D. MacMahon. 1988. Apparent heat increment and feeding strategy in walleye, *Stizostedion vitreum vitreum*. *Aquaculture* 68: 73-82.

- Beauchamp, D.A., D.J. Stewart, and G.L. Thomas. 1989. Corroboration of a bioenergetics model for sockeye salmon. *Transactions of the American Fisheries Society* 118: 597-607.
- Benjakul, S., T.A. Seymour, M.T. Morrissey, and H. An. 1997. Physicochemical changes in Pacific whiting muscle proteins during iced storage. *Journal of Food Science* 62: 729-733.
- Benoit, J.M., R.P. Mason, and C.C. Gilmore. 1999a. Estimation of mercury-sulfide speciation in sediment pore waters using octanol-water partitioning and implications for availability to methylating bacteria. *Environmental Toxicology and Chemistry* 18: 2138-2141.
- Benoit, J.M., C.C. Gilmore, R.P. Mason, and A. Heyes. 1999b. Sulfide controls on mercury speciation and bioavailability to methylating bacteria in sediment pore waters. *Environmental Science and Technology* 33: 951-957.
- Bergman, E. and L.A. Greenberg. 1994. Competition between a planktivore, a benthivore, and a species with ontogenetic diet shifts. *Ecology* 75: 1233-1245.
- Bettoli, P.W., M.J. Maceina, R.L. Noble, and R.K. Betsill. 1993. Response of a reservoir fish community to aquatic vegetation removal. *North American Journal of Fisheries Management* 13: 110-124.
- Binkowski, F.P. and L.G. Rudstam. 1994. Maximum daily ration of Great Lakes bloater. *Transactions of the American Fisheries Society* 123: 335-343.
- Bohlin, T., C. Dellefors, U. Faremo, and A. Johlander. 1994. The energetic equivalence hypothesis and the relation between population density and body size in stream-living salmonids. *American Naturalist* 143: 478-493.
- Bonner, T.H. and G.R. Wilde. 2002. Effects of turbidity on prey consumption of prairie stream fishes. *Transactions of the American Fisheries Society* 131: 1203-1208.
- Boudreau, P.R. and L.M. Dickie. 1989. Biological model of fisheries production based on physiological and ecological scalings of body size. *Canadian Journal of Fisheries and Aquatic Sciences* 46: 614-623.
- Boush, G.M. and J.R. Thieleke. 1983a. Mercury content in sharks. *Bulletin of Environmental Contamination and Toxicology* 30: 284-290.
- Boush, G.M. and J.R. Thieleke. 1983b. Total mercury content in yellowfin and bigeye tuna. *Bulletin of Environmental Contamination and Toxicology* 30: 291-297.
- Boxall, A., C. Brown, and K. Barrett. 2001. Higher tier laboratory aquatic toxicity testing. Cranfield University, Cranfield Centre for EcoChemistry, Silsoe, Beds MK45 4DT, UK. Research Report No. JF 4317E for DETR.
- Boxall, A.B.A., C.D. Brown, and K.L. Barrett. 2002. Higher-tier laboratory methods for assessing the aquatic toxicity of pesticides. *Pest Management Science* 58: 637-648.
- Bozek, M.A., T.M. Burri, and R.V. Frie. 1999. Diets of muskellunge in northern Wisconsin lakes. *North American Journal of Fisheries Management* 19: 258-270.
- Branson, D.R., I.T. Takahashi, W.M. Parker, and G.E. Blau. 1985. Bioconcentration kinetics of 2, 3, 7, 8 tetrachlorodibenzo-p-dioxin in rainbow trout. *Environmental Toxicology and Chemistry* 4: 779-788.
- Braune, B.M. 1987. Mercury accumulation in relation to size and age of Atlantic herring (*Clupea harengus harengus*) from the southwestern Bay of Fundy, Canada. *Archives of Environmental Contamination and Toxicology* 16: 311-320.
- Breck, J.E. and M.J. Gitter. 1983. Effect of fish size on the reactive distance of bluegill (*Lepomis macrochirus*) sunfish. *Canadian Journal of Fisheries and Aquatic Sciences* 40: 162-167.
- Brett, J.R. 1971. Satiation time, appetite, and maximum food intake of sockeye salmon (*Oncorhynchus nerka*). *Journal of the Fisheries Research Board of Canada* 28: 409-415.
- Brett, J.R. and C.A. Zala. 1975. Daily pattern of nitrogen excretion and oxygen consumption of sockeye salmon (*Oncorhynchus nerka*) under controlled conditions. *Journal of the Fisheries Research Board of Canada* 32: 2479-2486.
- Brett, J.R., J.E. Shelbourn, and C.T. Shoop. 1969. Growth rate and body composition of fingerling sockeye salmon, *Oncorhynchus nerka*, in relation to temperature and ration size. *Journal of the Fisheries Research Board of Canada* 26: 2363-2394.
- Briggs, G.G. 1981. Theoretical and experimental relationships between soil adsorption, octanol-water partition coefficients, water solubilities, bioconcentration factors, and the parachor. *Journal of Agricultural and Food Chemistry* 29: 1050-1059.
- Brink, F. and J.M. Posternak. 1948. Thermodynamic analysis of the relative effectiveness of narcotics. *Journal of Cellular and Comparative Physiology* 32: 211-233.

- Brockway, D.L., P.D. Smith, and M.C. Barber. 1996. PCBs in the Aquatic-Riparian zone of the Lake Hartwell Ecosystem, South Carolina. U.S. Environmental Protection Agency, National Exposure Research Laboratory, Athens, GA. Internal Report.
- Broderius, S. and M. Kahl. 1985. Acute toxicity of organic chemical mixtures to the fathead minnow. *Aquatic Toxicology* 6: 307-322.
- Brodeur, R.D. 1991. Ontogenetic variations in the type and size of prey consumed by juvenile coho, *Oncorhynchus kisutch*, and chinook, *O. tshawytscha*, salmon. *Environmental Biology of Fishes* 30: 303-315.
- Bromley, P.J. 1980. The effect of dietary water content and feeding rate on the growth and food conversion efficiency of turbot (*Scophthalmus maximus* L.). *Aquaculture* 20: 91-99.
- Brown, D.A. and T.R. Parsons. 1978. Relationship between cytoplasmic distribution of mercury and toxic effects to zooplankton and chum salmon (*Oncorhynchus keta*) exposed to mercury in a controlled ecosystem. *Journal of the Fisheries Research Board of Canada* 35: 880-884.
- Brown, G.M. 1960. Heat or mass transfer in a fluid in laminar flow in a circular or flat conduit. *AIChE Journal* 6: 179-183.
- Bruggeman, W.A., L.B.J.M. Martron, D. Kooiman, and O. Hutzinger. 1981. Accumulation and elimination kinetics of di-, tri-, and tetra chlorobiphenyls by goldfish after dietary and aqueous exposure. *Chemosphere* 10: 811-832.
- Buttkus, H. 1967. Amino acid composition of myosin from trout muscle. *Journal of the Fisheries Research Board of Canada* 24: 1607-1612.
- Buttkus, H. 1971. The sulfhydryl content of rabbit and trout myosins in relation to protein stability. *Canadian Journal of Biochemistry* 49: 97-107.
- Buynak, G.L., L.E. Kornman, A. Surmont, and B. Mitchell. 1991. Evaluation of a smallmouth bass stocking program in a Kentucky Stream. *North American Journal of Fisheries Management* 11: 293-297.
- Calamari, D. and J.S. Alabaster. 1980. An approach to theoretical models in evaluating the effects of mixtures of toxicants in the aquatic environment. *Chemosphere* 9: 533-538.
- Campfens, J. and D. Mackay. 1997. Fugacity-based model of PCB bioaccumulation in complex aquatic food webs. *Environmental Science and Technology* 31: 577-583.
- Carlander, K.D. 1969. Handbook of Freshwater Fishery Biology. Volume One: Life History Data on Freshwater Fishes of the United States and Canada, Exclusive of the Perciformes. Iowa State University Press, Ames, IA.
- Carreira, L.A., S. Hilal, and S.W. Karickhoff. 1994. Estimation of Chemical Reactivity Parameters and Physical Properties of Organic Molecules Using SPARC. *In* Theoretical and Computational Chemistry, Quantitative Treatment of Solute/Solvent Interactions. Edited by P. Politzer and J.S. Murray. Elsevier Publishers,
- Chapra, S.C. and J.M. Boyer. 1992. Fate of environmental pollutants. *Water Environmental Research* 64: 581-593.
- Chiou, C.T., R.L. Malcolm, T.I. Brinton, and D.E. Kile. 1986. Water solubility enhancement of some organic pollutants and pesticides by dissolved humic and fulvic acids. *Environmental Science and Technology* 20: 502-508.
- Chung, Y.-C., M.-L. Ho, F.-L. Chyan, and S.-T. Jiang. 2000. Utilization of freeze-dried mackerel (*Scomber australasicus*) muscle proteins as a binder in restructured meat. *Fisheries Science* 66: 130-135.
- Clark, K.E., F.A.P.C. Gobas, and D. Mackay. 1990. Model of organic chemical uptake and clearance by fish from food and water. *Environmental Science and Technology* 24: 1203-1213.
- Cohen, Y. and E.J. Cooter. 2002a. Multimedia Environmental Distribution of Toxics (Mend-Tox).II: Software Implementation and Case Studies. *Practice Periodical of Hazardous, Toxic, and Radioactive Waste Management* 6: 87-101.
- Cohen, Y. and E.J. Cooter. 2002b. Multimedia Environmental Distribution of Toxics (Mend-Tox).I: Hybrid compartmental-spatial modeling framework. *Practice Periodical of Hazardous, Toxic, and Radioactive Waste Management* 6: 70-86.
- Colton, C.K., K.A. Smith, P. Stroeve, and E.W. Merrill. 1971. Laminar flow mass transfer in a flat duct with permeable walls. *AIChE Journal* 17: 773-780.
- Connell, D. and R. Markwell. 1992. Mechanism and prediction of nonspecific toxicity to fish using bioconcentration characteristics. *Ecotoxicology and Environmental Safety* 24: 247-265.
- Connell, J.J. and P.F. Howgate. 1959. Studies on the proteins of fish skeletal muscle 6. Amino acid composition of cod fibrillar proteins. *Biochemical Journal* 71: 83-86.

- Connolly, J.P. and C.J. Pedersen. 1988. A thermodynamic-based evaluation of organic chemical accumulation in aquatic organisms. *Environmental Science and Technology* 22: 99-103.
- Cowan, C.E., D.J. Versteeg, R.J. Larson, and P.J. Kloepper-Sams. 1995. Integrated approach for environmental assessment of new and existing substances. *Regulatory Toxicology and Pharmacology* 21: 3-31.
- Cox, D.K. and C.C. Coutant. 1981. Growth dynamics of juvenile striped bass as functions of temperature and ration. *Transactions of the American Fisheries Society* 110: 226-238.
- Craig, J.F. 1977. The body composition of the adult perch, *Perca fluviatilis* in Windermere, with reference to seasonal changes and reproduction. *Journal of Animal Ecology* 46: 617-632.
- Cross, F.A., L.H. Hardy, N.Y. Jones, and R.T. Barber. 1973. Relation between total body weight and concentrations of manganese, iron, copper, zinc, and mercury in white muscle of bluefish (*Pomatomus saltatrix*) and a bathyl-demersal fish *Antimora rostrata*. *Journal of the Fisheries Research Board of Canada* 30: 1287-1291.
- Cuenca, M.L., R.R. Stickney, and W.E. Grant. 1985. Fish bioenergetics and growth in aquaculture ponds: I. individual fish model development. *Ecological Modelling* 27: 169-190.
- Dabrowska, H., S.W. Fisher, K. Dabrowska, and A.E. Staubus. 1996. Dietary uptake efficiency of HCBP in channel catfish: the effect of fish contaminant body burden. *Environmental Toxicology and Chemistry* 15: 746-749.
- Damuth, J. 1981. Population density and body size in mammals. *Nature (London)* 290: 699-700.
- Davis, J.A. and C.E. Boyd. 1978. Concentrations of selected elements and ash in bluegill (*Lepomis macrochirus*) and certain other freshwater fish. *Transactions of the American Fisheries Society* 107: 862-867.
- Dawson, D.A. 1994. Joint action of carboxyl acid binary mixtures on *Xenopus* embryo development: comparison of joint actions for malformation types. *Archives of Environmental Contamination and Toxicology* 27: 243-249.
- de Bruijn, J., W. Seinen, and J. Hermens. 1993. Biotransformation of organophosphorus compounds by rainbow trout (*Oncorhynchus mykiss*) liver in relation to bioconcentration. *Environmental Toxicology and Chemistry* 12: 1041-1050.
- de Robertis, A., C.H. Ryer, A. Veloza, and R.D. Brodeur. 2003. Differential effects of turbidity on prey consumption of piscivorous and planktivorous fish. *Canadian Journal of Fisheries and Aquatic Sciences* 60: 1517-1526.
- de Wolf, W., J.H.M. de Bruijn, W. Seinen, and J.L.M. Hermens. 1992. Influence of biotransformation on the relationship between bioconcentration factors and octanol-water partition coefficients. *Environmental Science and Technology* 26: 1197-1201.
- Deliman, P.N. and J.A. Gerald. 1998. Development of a Multimedia Exposure Assessment Model for Evaluating Ecological Risk of Exposure to Military-Related Compounds (MRCs) at Military Sites. US Army Corps of Engineers, Waterways Experiment Station, Wahington, DC. Technical Report IRRP-98-9.
- Dennis, J.E., D.M. Gay, and R.E. Welsh. 1981. Algorithm 573 NL2SOL: An Adaptive Nonlinear Least-squares Algorithm. *ACM Transactions on Mathematical Software* 7: 369-383.
- Dettmers, J.M., S. Gutreuter, D.H. Wahl, and D.A. Soluk. 2001. Patterns in abundance of fishes in main channels of the upper Mississippi River system. *Canadian Journal of Fisheries and Aquatic Sciences* 58: 933-942.
- Dickie, L.M., S.R. Kerr, and P.R. Boudreau. 1987. Size-dependent processes underlying regularities in ecosystem structure. *Ecological Monographs* 57: 233-250.
- Dixon, K.R. and J.D. Florian, Jr. 1993. Modeling mobility and effects of contaminants in wetlands. *Environmental Toxicology and Chemistry* 12: 2281-2292.
- Doi, A.M., Z. Lou, E. Holmes, C.-L.J. Li, C.S. Venugopal, M.O. James, and K.M. Kleinow. 2000. Effects of micelle fatty acid composition and 3, 4, 3', 4'-tetrachlorobiphenyl (TCB) exposure on intestinal [¹⁴C]-TCB bioavailability and biotransformation in channel catfish in situ preparations. *Toxicological Sciences* 55: 85-96.
- Donkin, P., J. Widdows, S.V. Evans, C.M. Worrall, and M. Carr. 1989. Quantitative structure-activity relationships for the effect of hydrophobic organic chemicals on rate of feeding by mussels (*Mytilus edulis*). *Aquatic Toxicology* 14: 277-294.
- Driscoll, S.K. and P.F. Landrum. 1997. A comparison of equilibrium partitioning and critical body residue approaches for predicting toxicity of sediment-associated fluoranthene to freshwater amphipods. *Environmental Toxicology and Chemistry* 16: 2179-2186.

- du Preez, H.H. and A.C. Cockroft. 1988a. Nonfaecal and faecal losses of *Pomadasys commersonni* (Teleostei: Pomadasyidae) feeding on the surf clam, *Donax serra*. *Comparative Biochemistry and Physiology A* 90: 63-70.
- du Preez, H.H. and A.C. Cockroft. 1988b. Nonfaecal and faecal losses of the marine teleost, *Lichia amia* (Linnaeus, 1758), feeding on live southern mullet, *Liza richardsonii* (Smith, 1846). *Comparative Biochemistry and Physiology A* 90: 71-77.
- Dunbrack, R.L. 1988. Feeding of juvenile coho salmon (*Oncorhynchus kisutch*): maximum appetite, sustained feeding rate, appetite return, and body size. *Canadian Journal of Fisheries and Aquatic Sciences* 45: 1191-1196.
- Dunham, J.B. and G.L. Vinyard. 1997. Relationships between body mass, population density, and the self-thinning rule in stream-living salmonids. *Canadian Journal of Fisheries and Aquatic Sciences* 54: 1025-1030.
- Dunham, J.B., B.R. Dickerson, E. Beever, R.D. Duncan, and G.L. Vinyard. 2000. Effects of food limitation and emigration on self-thinning in experimental minnow cohorts. *Journal of Animal Ecology* 69: 927-934.
- Dutton, M.D., M. Stephenson, and J.F. Klaverkamp. 1993. A mercury saturation assay for measuring metallothionein in fish. *Environmental Toxicology and Chemistry* 12: 1193-1202.
- East, P. and P. Magnan. 1991. Some factors regulating piscivory of brook trout, *Salvelinus fontinalis*, in lakes of the Laurentian Shield. *Canadian Journal of Fisheries and Aquatic Sciences* 48: 1735-1743.
- ECOFRAM. 1999. Aquatic Draft Report (<http://www.epa.gov/oppefed1/ecorisk/aquareport.pdf>). Ecological Committee on Federal, Insecticide, Fungicide and Rodenticide Act, Risk Assessment.
- ECOFRAM Aquatic Effects Subcommittee, J. Giddings, L. Barnhouse, D. Farrar, T. Hall, M. McKee, M. Newman, K. Reinert, B. Sebastien, K. Solomon, A. Stavola, L. Touart, and R. Wentzel. 1998. Preliminary Findings of the Ecological Committee on Fifra Risk Assessment Methods (Ecofram): VI. Aquatic Effects Analysis. Poster (<http://www.epa.gov/oppefed1/ecorisk/setac98a.pdf>), SETAC Annual Meeting, Charlotte, NC.
- Elliott, J.M. 1975. Weight of food and time required to satiate brown trout, *Salmo trutta* L. *Freshwater Biology* 5: 51-64.
- Elliott, J.M. 1976a. Body composition of the brown trout (*Salmo trutta* L.) in relation to temperature and ration size. *Journal of Animal Ecology* 45: 273-289.
- Elliott, J.M. 1976b. The energetics of feeding, metabolism and growth of brown trout (*Salmo trutta* L.) in relation to body weight, water temperature and ration size. *Journal of Animal Ecology* 45: 923-948.
- Elliott, J.M. 1993. The self-thinning rule applied to juvenile sea-trout, *Salmo trutta*. *Journal of Animal Ecology* 62: 371-379.
- Elliott, J.M. and L. Persson. 1978. The estimation of daily rates of food consumption for fish. *Journal of Animal Ecology* 47: 977-991.
- Elliott, J.M. and M.A. Hurley. 1998. A new functional model for estimating the maximum amount of invertebrate food consumed per day by brown trout, *Salmo trutta*. *Freshwater Biology* 39: 339-349.
- Elrod, J.H. and R. O'Gorman. 1991. Diet of juvenile lake trout in southern Lake Ontario in relation to abundance and size of prey fishes, 1979-1987. *Transactions of the American Fisheries Society* 120: 290-302.
- Elston, D.A. 1992. Sensitivity analysis in the presence of correlated parameter estimates. *Ecological Modelling* 64: 11-22.
- Ensign, W.E., R.J. Strange, and S.E. Moore. 1990. Summer food limitation reduces brook and rainbow trout biomass in a Southern Appalachian stream. *Transactions of the American Fisheries Society* 119: 894-901.
- Erickson, R.J. and J.M. McKim. 1990. A model for exchange of organic chemicals at fish gills: flow and diffusion limitations. *Aquatic Toxicology* 18: 175-198.
- Eschmeyer, P.H. and A.M. Phillips, Jr. 1965. Fat content of the flesh of siscowets and lake trout from Lake Superior. *Transactions of the American Fisheries Society* 94: 62-74.
- Exponent. 1998. Review of Bioaccumulation Methods for Fish and Shellfish. Exponent, Bellevue, WA. Prepared for Washington State Department of Ecology, Olympia, WA.
- Exponent. 2003. Fish Contaminant Monitoring Program: Review and Recommendations. Exponent, Bellevue, WA. Prepared for Michigan Department of Environmental Quality, Water Division, Lansing, MI.
- Eyring, H., J.W. Woodbury, and J.S. D'Arrigo. 1973. A molecular mechanism of general anesthesia. *Journal of*

Anesthesiology 38: 415-424.

Facey, D.E. and G.D. Grossman. 1990. The metabolic cost of maintaining position for four North American stream fishes: effects of season and velocity. *Physiological Zoology* 63: 757-776.

Feijtel, T., P. Klopper-Sams, K. den Haan, R. van Egmond, M. Comber, R. Heusel, P. Wierich, W. Ten Berge, A. Gard, W. de Wolf, and H. Niessen. 1997. Integration of bioaccumulation in an environmental risk assessment. *Chemosphere* 34: 2337-2350.

Ferguson, J. 1939. Use of chemical potentials as indices of toxicity. *Proceedings of the Royal Society of London Series B - Biological Sciences* 127: 387-404.

Fisk, A.T., R.J. Norstrom, C.D. Cymbalisky, and D.C.G. Muir. 1998. Dietary accumulation and depuration of hydrophobic organochlorines: bioaccumulation parameters and their relationship with the octanol/water partition coefficient. *Environmental Toxicology and Chemistry* 17: 951-961.

Flath, L.E. and J.S. Diana. 1985. Seasonal energy dynamics of the alewife in southeastern Lake Michigan. *Transactions of the American Fisheries Society* 114: 328-337.

Flick, W.A. and D.A. Webster. 1992. Standing crop of brook trout in Adirondack waters before and after removal of non-trout species. *North American Journal of Fisheries Management* 12: 783-796.

Forrester, C.R., K.S. Ketchen, and C.C. Wong. 1972. Mercury content of spiny dogfish (*Squalus acanthias*) in the Strait of Georgia, British Columbia. *Journal of the Fisheries Research Board of Canada* 29: 1487-1490.

Franks, N.P. and W.R. Lieb. 1982. Molecular mechanisms of general anaesthesia. *Nature (London)* 300: 487-493.

Franks, N.P. and W.R. Lieb. 1984. Do general anaesthetics act by competitive binding to specific receptors? *Nature (London)* 310: 599-601.

Friant, S.L. and L. Henry. 1985. Relationship between toxicity of certain organic compounds and their concentrations in tissues of aquatic organisms: a perspective. *Chemosphere* 14: 1897-1907.

Fulton, C.T. 1977. Two-point boundary value problems with eigenvalue parameter contained in the boundary conditions. *Proceedings of the Royal Society of Edinburgh Section A-mathematics* 77: 293-308.

Galis, F. and C.D.N. Barel. 1980. Comparative functional morphology of the gills of African lacustrine Cichlidae (Pisces, Teleostei). *Netherlands Journal of Zoology* 30: 392-430.

Gallagher, M.L., E. Kane, and J. Courtney. 1984. Differences in oxygen consumption and ammonia production among American elvers (*Anguilla rostrata*). *Aquaculture* 40: 183-187.

Garber, K.J. 1983. Effect of fish size, meal size and dietary moisture on gastric evacuation of pelleted diets by yellow perch, *Perca flavescens*. *Aquaculture* 34: 41-49.

Garman, G.C. and L.A. Nielsen. 1982. Piscivory by stocked brown trout (*Salmo trutta*) and its impact on the nongame fish community of Bottom Creek, Virginia. *Canadian Journal of Fisheries and Aquatic Sciences* 39: 862-869.

Giersch, C. 1991. Sensitivity analysis of ecosystems: an analytical treatment. *Ecological Modelling* 53: 131-146.

Gillen, A.L., R.A. Stein, and R.F. Carline. 1981. Predation by pellet-reared tiger muskellunge on minnows and bluegills in experimental systems. *Transactions of the American Fisheries Society* 110: 197-209.

Gobas, F.A.P.C. 1993. A model for predicting the bioaccumulation of hydrophobic organic chemicals in food-webs: application to Lake Ontario. *Ecological Modelling* 69: 1-17.

Gobas, F.A.P.C. and D. Mackay. 1987. Dynamics of hydrophobic organic chemical bioconcentration in fish. *Environmental Toxicology and Chemistry* 6: 495-504.

Gobas, F.A.P.C. and H.A. Morrison. 2000. Chapter 9. Bioconcentration and Biomagnification in the Aquatic Environment. *In Handbook of Property Estimation Methods for Chemicals: Environmental and Health Sciences. Edited by R.S. Boethling and D. Mackay. Lewis Publishers, CRC Press, Boca Raton, FL.*

Gobas, F.A.P.C., A. Opperhuizen, and O. Hutzinger. 1986. Bioconcentration of hydrophobic chemicals in fish: relationship with membrane permeation. *Environmental Toxicology and Chemistry* 5: 637-646.

Gobas, F.A.P.C., D.C.G. Muir, and D. Mackay. 1988. Dynamics of dietary bioaccumulation and faecal elimination of hydrophobic organic chemicals in fish. *Chemosphere* 17: 943-962.

Gobas, F.A.P.C., J.R. McCorquodale, and G.D. Haffner. 1993. Intestinal absorption and biomagnification of organochlorines. *Environmental Toxicology and Chemistry* 12: 567-576.

- Gordoa, A. and C.M. Duarte. 1992. Size-dependent density of the demersal fish off Namibia: patterns within and among species. *Canadian Journal of Fisheries and Aquatic Sciences* 49: 1990-1993.
- Grabner, M. and R. Hofer. 1985. The digestibility of the proteins of broad bean (*Vicia faba*) and soya bean (*Glycine max*) under in vitro conditions simulating the alimentary tracts of rainbow trout (*Salmo gairdneri*) and carp (*Cyprinus carpio*). *Aquaculture* 48: 111-122.
- Grant, J.W.A. and D.L. Kramer. 1990. Territory size as a predictor of the upper limit to population density of juvenile salmonids in streams. *Canadian Journal of Fisheries and Aquatic Sciences* 47: 1724-1737.
- Grant, J.W.A., S.O. Steingrímsson, E.R. Keeley, and R.A. Cunjak. 1998. Implications of territory size for the measurement and prediction of salmonid abundance in streams. *Canadian Journal of Fisheries and Aquatic Sciences* 55 (Suppl. 1): 181-190.
- Gregory, R.S. 1993. Effect of turbidity on the predator avoidance behavior of juvenile chinook salmon (*Oncorhynchus tshawytscha*). *Canadian Journal of Fisheries and Aquatic Sciences* 50: 241-246.
- Gregory, R.S. and T.G. Northcote. 1993. Surface, planktonic, and benthic foraging by juvenile chinook salmon (*Oncorhynchus tshawytscha*) in turbid laboratory conditions. *Canadian Journal of Fisheries and Aquatic Sciences* 50: 233-240.
- Grieb, T.M., C.T. Driscoll, S.P. Gloss, C.L. Schofield, G.L. Bowie, and D.B. Porcella. 1990. Factors affecting mercury accumulation in fish in the upper Michigan peninsula. *Environmental Toxicology and Chemistry* 9: 919-930.
- Grimsrud, L. and A.L. Babb. 1966. Velocity and concentration profiles for laminar flow of a Newtonian fluid in a dialyzer. *Chemical Engineering Progress, Symposium Series* (66) 62: 19-31.
- Groves, T.D.D. 1970. Body composition changes during growth in young sockeye (*Oncorhynchus nerka*) in fresh water. *Journal of the Fisheries Research Board of Canada* 27: 929-942.
- Gruger, E.H., Jr, N.L. Karrick, A.I. Davidson, and T. Hruby. 1975. Accumulation of 3, 4, 3', 4'-tetrachlorobiphenyl and 2, 4, 5, 2', 4', 5'-hexachlorobiphenyl in juvenile coho salmon. *Environmental Science and Technology* 9: 121-127.
- Grzenda, A.R., D.F. Paris, and W.J. Taylor. 1970. The uptake, metabolism, and elimination of chlorinated residues by goldfish (*Carassius auratus*) fed a ¹⁴C-DDT contaminated diet. *Transactions of the American Fisheries Society* 99: 385-396.
- Guíñez, R. 2005. A review on self-thinning in mussels. *Revista de Biología Marina y Oceanografía* 40: 1- 6.
- Gutenmann, W.H., J.G. Ebel, Jr, H.T. Kuntz, K.S. Yourstone, and D.J. Lisk. 1992. Residues of p, p'-DDE and mercury in lake trout as a function of age. *Archives of Environmental Contamination and Toxicology* 22: 452-455.
- Habera, J.W., R.D. Bivens, B.D. Carter, and C.E. Williams. 2004. Trout Fisheries Report Region IV 2003. Tennessee Wildlife Resources Agency, Nashville, TN. Fisheries Report No. 04-04.
- Håkanson, L. 1995. Optimal size of predictive models. *Ecological Modelling* 78: 195-204.
- Hale, R.S. 1996. Threadfin shad use as supplemental prey in reservoir white crappie fisheries in Kentucky. *North American Journal of Fisheries Management* 16: 619-632.
- Hallam, T.G. and Q. Deng. 2006. Simulation of structured populations in chemically stressed environments. *Mathematical Biosciences and Engineering* 3: 51-65.
- Hambright, K.D. 1991. Experimental analysis of prey selection by largemouth bass: role of predator mouth width and prey body depth. *Transactions of the American Fisheries Society* 120: 500-508.
- Hamilton, S.J. and P.M. Mehrle. 1986. Metallothionein in fish: review of its importance in assessing stress from metal contaminants. *Transactions of the American Fisheries Society* 115: 596-609.
- Hamilton, S.J., P.M. Mehrle, and J.R. Jones. 1987a. Cadmium-saturation technique for measuring metallothionein in brook trout. *Transactions of the American Fisheries Society* 116: 541-550.
- Hamilton, S.J., P.M. Mehrle, and J.R. Jones. 1987b. Evaluation of metallothionein measurement as a biological indicator of stress from cadmium in brook trout. *Transactions of the American Fisheries Society* 116: 551-560.
- Hanson, D. and K. Johansen. 1970. Relationship of gill ventilation and perfusion in Pacific dogfish, *Squalus suckleyi*. *Journal of the Fisheries Research Board of Canada* 27: 551-564.

- Hanson, J.M. and W.C. Leggett. 1986. Effect of competition between two freshwater fishes on prey consumption and abundance. *Canadian Journal of Fisheries and Aquatic Sciences* 43: 1363-1372.
- Hanson, P.C., T.B. Johnson, D.E. Schindler, and J.F. Kitchell. 1997. *Fish Bioenergetics 3.0 for Windows*. University of Wisconsin, Sea Grant Institute, Madison, WI. U.S. Department of Commerce, National Technical Information Service PB98-502370.
- Hartman, K.J. and S.B. Brandt. 1993. Systematic sources of bias in a bioenergetics model: Example for age-0 striped bass. *Transactions of the American Fisheries Society* 122: 912-926.
- Hartman, K.J. and S.B. Brandt. 1995a. Comparative energetics and development of bioenergetic models for sympatric estuarine piscivores. *Canadian Journal of Fisheries and Aquatic Sciences* 52: 1647-1666.
- Hartman, K.J. and S.B. Brandt. 1995b. Estimating energy density of fish. *Transactions of the American Fisheries Society* 124: 347-355.
- Haydon, D.A., B.M. Hendry, and S.R. Levinson. 1977. The molecular mechanisms of anaesthesia. *Nature (London)* 268: 356-358.
- Hayduk, W. and H. Laudie. 1974. Prediction of diffusion coefficients for nonelectrolytes in dilute aqueous solutions. *AIChE Journal* 20: 611-615.
- Hayward, R.S. and E. Arnold. 1996. Temperature dependence of maximum daily consumption in white crappie: implications for fisheries management. *Transactions of the American Fisheries Society* 125: 132-138.
- Hermanutz, R.O., J.G. Eaton, and L.H. Muller. 1985. Toxicity of endrin and malathion mixtures to flagfish (*Jordanella floridae*). *Archives of Environmental Contamination and Toxicology* 14: 307-314.
- Hermens, J. and P. Leeuwangh. 1982. Joint toxicity of mixtures of 8 and 24 chemicals to the guppy (*Poecilia reticulata*). *Ecotoxicology and Environmental Safety* 6: 302-310.
- Hermens, J., P. Leeuwangh, and A. Musch. 1985a. Joint toxicity of mixtures of groups of organic aquatic pollutants to the guppy (*Poecilia reticulata*). *Ecotoxicology and Environmental Safety* 9: 321-326.
- Hermens, J., H. Canton, P. Janssen, and R. de Jong. 1984a. Quantitative structure-activity relationships and toxicity studies of mixtures of chemicals with anaesthetic potency: acute lethal and sublethal toxicity to *Daphnia magna*. *Aquatic Toxicology* 5: 143-154.
- Hermens, J., H. Canton, N. Steyger, and R. Wegman. 1984b. Joint effects of a mixture of 14 chemicals on mortality and inhibition of reproduction of *Daphnia magna*. *Aquatic Toxicology* 5: 315-322.
- Hermens, J., E. Broekhuizen, H. Canton, and R. Wegman. 1985b. Quantitative structure activity relationships and mixture toxicity studies of alcohols and chlorohydrocarbons: effects on growth of *Daphnia magna*. *Aquatic Toxicology* 6: 209-217.
- Hermens, J., H. Könemann, P. Leeuwangh, and A. Musch. 1985c. Quantitative structure-activity relationships in aquatic toxicity studies of chemicals and complex mixtures of chemicals. *Environmental Toxicology and Chemistry* 4: 273-279.
- Hills, B.A. and G.M. Hughes. 1970. A dimensional analysis of oxygen transfer in fish gills. *Respiratory Physiology* 9: 126-140.
- Holling, C.S. 1966. The functional response of invertebrate predators to prey density. *Memoirs of the Entomological Society of Canada* 48: 1-86.
- Howgate, P. 1998. Review of the public health safety of products from aquaculture. *International Journal of Food Science and Technology* 33: 99-125.
- Hughes, G.M. 1966. The dimensions of fish gills in relation to their function. *Journal of Experimental Biology* 45: 177-195.
- Hughes, G.M., S.F. Perry, and J. Piiper. 1986. Morphometry of the gills of the elasmobranch *Scyliorhinus stellaris* in relation to body size. *Journal of Experimental Biology* 121: 27-42.
- Hunt, T., K. Roberts, and K. Red Knox. 1992. Bioaccumulation simulation of DDT by channel catfish, 13th Annual Meeting Society of Environmental Toxicology and Chemistry.
- Ingram, W. and C.D. Ziebell. 1983. Diet shifts to benthic feeding by threadfin shad. *Transactions of the American Fisheries Society* 112: 554-556.
- Itano, K. and K. Sasaki. 1983. Binding of mercury compounds to protein components of fish muscle. *Bulletin of the Japanese Society of Scientific Fisheries* 49: 1849-1853.
- Itoh, Y., R. Yoshinaka, and S. Ikeda. 1979. Effects of sulfhydryl reagents on the gel formation of carp actomyosin by heating.

- Bulletin of the Japanese Society of Scientific Fisheries 45: 1023-1025.
- Jackson, L.J. 1996. A simulation model of PCB dynamics in the Lake Ontario pelagic food web. *Ecological Modelling* 93: 43-56.
- Janes, N. and R.C. Playle. 1995. Modeling silver binding to gills of rainbow trout (*Oncorhynchus mykiss*). *Environmental Toxicology and Chemistry* 14: 1847-1858.
- Janoff, A.S., M.J. Pringle, and K.W. Miller. 1981. Correlation of general anesthetic potency with solubility in membranes. *Biochimica et Biophysica Acta* 649: 125-128.
- Jensen, A.L. 1986. Functional regression and correlation analysis. *Canadian Journal of Fisheries and Aquatic Sciences* 43: 1742-1745.
- Jensen, A.L., S.A. Spigarelli, and M.M. Thommes. 1982. PCB uptake by five species of fish in Lake Michigan, Green Bay of Lake Michigan, and Cayuga Lake, New York. *Canadian Journal of Fisheries and Aquatic Sciences* 39: 700-709.
- Jobling, M. 1981. Mathematical models of gastric emptying and the estimation of daily rates of food consumption for fish. *Journal of Fish Biology* 19: 245-257.
- Jobling, M. 1986. Mythical models of gastric emptying and implications for food consumption studies. *Environmental Biology of Fishes* 16: 35-50.
- Jobling, M. 1987. Influence of particle size and dietary energy content on patterns of gastric evacuation in fishes: test of a physiological model of gastric emptying. *Journal of Fish Biology* 30: 299-314.
- Johnson, B.L., D.L. Smith, and R.F. Carline. 1988. Habitat preferences, survival, growth, foods, and harvests of walleye and walleye × sauger hybrids. *North American Journal of Fisheries Management* 8: 292-304.
- Johnson, P.C. and G.L. Vinyard. 1987. Filter-feeding behavior and particle retention efficiency of Sacramento Blackfish. *Transactions of the American Fisheries Society* 116: 634-640.
- Joiris, C.R., I.B. Ali, L. Holsbeck, M. Bossicart, and G. Tapia. 1995. Total and organic mercury in Barents Sea pelagic fish. *Bulletin of Environmental Contamination and Toxicology* 55: 674-681.
- Jones, K.C., T. Keating, P. Diage, and A.C. Chang. 1991. Transport and food chain modeling and its role in assessing human exposure to organic chemicals. *Journal of Environmental Quality* 20: 317-329.
- Juanes, F. 1986. Population density and body size in birds. *American Naturalist* 128: 921-929.
- Juanes, F., R.E. Marks, K.A. McKown, and D.O. Conover. 1993. Predation by age-0 bluefish on age-0 anadromous fishes in the Hudson River estuary. *Transactions of the American Fisheries Society* 122: 348-356.
- Jude, D.J., F.J. Tesar, S.F. Deboe, and T.J. Miller. 1987. Diet and selection of major prey species by Lake Michigan salmonines, 1973-1982. *Transactions of the American Fisheries Society* 116: 677-691.
- Karickhoff, S.W. 1981. Semi-empirical estimation of sorption of hydrophobic pollutants on natural sediments and soil. *Chemosphere* 10: 833-846.
- Keeley, E.R. 2003. An experimental analysis of self-thinning in juvenile steelhead trout. *Oikos* 102: 543-550.
- Keeley, E.R. and J.W.A. Grant. 1995. Allometric and environmental correlates of territory size in juvenile Atlantic salmon (*Salmo salar*). *Canadian Journal of Fisheries and Aquatic Sciences* 52: 186-196.
- Kitchell, J.F., D.J. Stewart, and D. Weininger. 1977. Application of a bioenergetics model to yellow perch (*Perca flavescens*) and walleye (*Stizostedion vitreum vitreum*). *Journal of the Fisheries Research Board of Canada* 34: 1922-1935.
- Klaverkamp, J.F. and D.A. Duncan. 1987. Acclimation to cadmium toxicity by white suckers: cadmium binding capacity and metal distribution in gill and liver cytosol. *Environmental Toxicology and Chemistry* 6: 275-289.
- Kleeman, J.M., J.R. Olson, S.M. Chen, and R.E. Peterson. 1986a. Metabolism and deposition of 2, 3, 7, 8-tetrachlorodibenzo-p-dioxin in rainbow trout. *Toxicology and Applied Pharmacology* 83: 391-401.
- Kleeman, J.M., J.R. Olson, S.M. Chen, and R.E. Peterson. 1986b. Metabolism and deposition of 2, 3, 7, 8-tetrachlorodibenzo-p-dioxin in yellow perch. *Toxicology and Applied Pharmacology* 83: 402-411.
- Knight, R.L., F.J. Margraf, and R.F. Carline. 1984. Piscivory by walleyes and yellow perch in western Lake Erie. *Transactions of the American Fisheries Society* 113: 677-693.

- Knight, W. 1968. Asymptotic growth: An example of nonsense disguised as mathematics. *Journal of the Fisheries Research Board of Canada* 25: 1303-1307.
- Knouft, J.H. 2002. Regional analysis of body size and population density in stream fish assemblages: testing predictions of the energetic equivalence rule. *Canadian Journal of Fisheries and Aquatic Sciences* 59: 1350-1360.
- Könemann, H. 1981a. Quantitative structure-activity relationships in fish toxicity studies. Part 1: relationship for 50 industrial pollutants. *Toxicology* 19: 209-221.
- Könemann, H. 1981b. Fish toxicity tests with mixtures of more than two chemicals: a proposal for a quantitative approach and experimental results. *Toxicology* 19: 229-238.
- Kunisaki, N., K. Takada, and H. Matsuura. 1986. On the study of lipid content, muscle hardness, and fatty acid composition of wild and cultured horse mackerel. *Nippon Suisan Gakkaishi* 52: 333-336.
- Kuns, M.M. and W.G. Sprules. 2000. Zooplankton production in Lake Ontario: a multistrata approach. *Canadian Journal of Fisheries and Aquatic Sciences* 57: 2240-2247.
- Kushlan, J.A., S.A. Voorhees, W.F. Loftus, and P.C. Frohring. 1986. Length, mass, and calorific relationships of Everglades animals. *Florida Scientist* 49: 65-79.
- Lacasse, S. and P. Magnan. 1992. Biotic and abiotic determinants of the diet of brook trout, *Salvelinus fontinalis*, in lakes of the Laurentian Shield. *Canadian Journal of Fisheries and Aquatic Sciences* 49: 1001-1009.
- Lammens, E.H.R.R., H.W. de Nie, and J. Vijverberg. 1985. Resource partitioning and niche shifts of bream (*Abramis brama*) and eel (*Anguilla anguilla*) mediated by predation of smelt (*Osmerus eperlanus*) on *Daphnia hyalina*. *Canadian Journal of Fisheries and Aquatic Sciences* 42: 1342-1351.
- Landrum, P.F., H. Lee, II, and M.J. Lydy. 1992. Toxicokinetics in aquatic systems: model comparisons and use in hazard assessment. *Environmental Toxicology and Chemistry* 11: 1709-1725.
- Lange, T.R., H.E. Royals, and L.L. Connor. 1993. Influence of water chemistry on mercury concentration in largemouth bass from Florida lakes. *Transactions of the American Fisheries Society* 122: 74-84.
- Lantry, B.F. and D.J. Stewart. 1993. Ecological energetics of rainbow smelt in the Laurentian Great Lakes: an interlake comparison. *Transactions of the American Fisheries Society* 122: 951-976.
- Larsson, S. and I. Berglund. 1998. Growth and consumption of 0+ Arctic charr fed pelleted or natural foods at six different temperatures. *Journal of Fish Biology* 52: 230-242.
- Lassiter, R.R. 1975. Modeling dynamics of biological and chemical components of aquatic ecosystems. U.S. Environmental Protection Agency, Office of Research and Development, National Environmental Research Center, Corvallis, OR. EPA/660/3-75-012.
- Lassiter, R.R. 1990. A theoretical basis for predicting sublethal effects of toxic chemicals. U.S. Environmental Protection Agency, Office of Research and Development, Athens, GA.
- Lassiter, R.R. and D.K. Kearns. 1974. Phytoplankton population changes and nutrient fluctuations in a simple ecosystem model. *In Modeling the Eutrophication Process. Edited by E.J. Middlebrooks, D.H. Falkenberg and T.E. Maloney.* Ann Arbor Scientific Publication, Ann Arbor, MI. pp. 131-138.
- Lassiter, R.R. and T.G. Hallam. 1990. Survival of the fattest: implications for acute effects of lipophilic chemicals on aquatic populations. *Environmental Toxicology and Chemistry* 9: 585-595.
- Law, R.H., A. Mellors, and F.R. Hallet. 1985. Physical aspects of inhibition of enzymes by hydrocarbons: inhibition of a-chymotrypsin by chlorinated aromatics and alkanes. *Environmental Research* 36: 193-205.
- Lewis, W.M., R. Heidinger., W. Kirk, W. Chapman, and D. Johnson. 1974. Food intake of the largemouth bass. *Transactions of the American Fisheries Society* 103: 277-280.
- Lieb, A.J., D.D. Bills, and R.O. Sinnhuber. 1974. Accumulation of dietary polychlorinated biphenyls (Aroclor 1254) by rainbow trout (*Salmo gairdneri*). *Journal of Agricultural and Food Chemistry* 22: 638-642.
- Lien, G.J. and J.M. McKim. 1993. Predicting branchial and cutaneous uptake of 2, 2', 5, 5'-tetrachlorobiphenyl in fathead minnows (*Pimephales promelas*) and Japanese medaka (*Oryzias latipes*): rate limiting factors. *Aquatic Toxicology* 27: 15-32.
- Lien, G.J., J.W. Nichols, J.M. McKim, and C.A. Gallinat. 1994. Modeling the accumulation of three waterborne chlorinated ethanes in fathead minnows (*Pimephales promelas*): a physiologically based approach. *Environmental Toxicology and*

Chemistry 13: 1195-1205.

Limno-Tech. 2002. Descriptive Inventory of Models with Prospective Relevance to Ecological Impacts of Water Withdrawals. Limno-Tech, Inc., Ann Arbor, MI. Prepared for The Great Lakes Commission, October 14, 2002.

Lin, T.M. and J.W. Park. 1998. Solubility of salmon myosin as affected by conformational changes at various ionic strengths and pH. *Journal of Food Science* 63: 215-218.

Liu, J., Y. Cui, and J. Liu. 1998. Food consumption and growth of two piscivorous fishes, the mandarin fish and the Chinese snakehead. *Journal of Fish Biology* 53: 1071-1083.

Lockhart, W.L., J.F. Uthe, A.R. Kenney, and P.M. Mehrle. 1972. Methylmercury in northern pike (*Esox lucius*): distribution, elimination, and some biochemical characteristics of contaminated fish. *Journal of the Fisheries Research Board of Canada* 29: 1519-1523.

Loehle, C. 1997. A hypothesis testing framework for evaluating ecosystem model performance. *Ecological Modelling* 97: 153-165.

Loftus, W.F., J.C. Trexler, and R.D. Jones. 1998. Mercury transfer through an Everglades aquatic food web. Florida Department of Environmental Protection. Final Report Contract SP-329.

Lorenzen, K. 1996. The relationship between body weight and natural mortality in juvenile and adult fish: a comparison of natural ecosystems and aquaculture. *Journal of Fish Biology* 49: 627-647.

Lowe, T.P., T.W. May, W.G. Brumbaugh, and D.A. Kane. 1985. National Contaminant Biomonitoring Program: Concentrations of seven elements in freshwater fish, 1979-1981. *Archives of Environmental Contamination and Toxicology* 14: 363-388.

Luk, G.K. and F. Brockway. 1997. Application of a polychlorinated biphenyls bioaccumulation model to Lake Ontario lake trout. *Ecological Modelling* 101: 97-111.

Luten, J.B., W. Bouquet, G. Riekwel-Booy, A.B. Rauchbaar, and M.W.M. Scholte. 1987. Mercury in flounder, *Platichthys flesus*, cod, *Gadus morhua*, and perch, *Perca fluviatilis*, in relation to their length and environment. *Bulletin of Environmental Contamination and Toxicology* 38: 318-323.

MacCrimmon, H.R., C.D. Wren, and B.L. Gots. 1983. Mercury uptake by lake trout, *Salvelinus namaycush*, relative to age,

growth, and diet in Tadenac Lake with comparative data from other PreCambrian Shield lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 40: 114-120.

Maceina, M.J., W.B. Wrenn, and D.R. Lowery. 1995. Estimating harvestable largemouth bass abundance in a reservoir with an electrofishing catch depletion technique. *North American Journal of Fisheries Management* 15: 103-109.

Mackay, D. 1982. Correlation of bioconcentration factors. *Environmental Science and Technology* 16: 274-278.

Mackay, D. and A.I. Hughes. 1984. Three-parameter equation describing the uptake of organic compounds by fish. *Environmental Science and Technology* 18: 430-444.

Mackay, D. and A. Fraser. 2000. Bioaccumulation of persistent organic chemicals: mechanisms and models. *Environmental Pollution* 110: 375-391.

MacRae, R.K., D.E. Smith, N. Swoboda-Colberg, J.S. Meyer, and H.L. Bergman. 1999. Copper binding affinity of rainbow trout (*Oncorhynchus mykiss*) and brook trout (*Salvelinus fontinalis*) gills: implications for assessing bioavailable metal. *Environmental Toxicology and Chemistry* 18: 1180-1189.

Madenjian, C.P., T.J. DeSorcie, and R.M. Stedman. 1998. Ontogenic and spatial patterns in diet and growth of lake trout in Lake Michigan. *Transactions of the American Fisheries Society* 127: 236-252.

Madenjian, C.P., S.R. Carpenter, G.E. Eck, and M.A. Miller. 1993. Accumulation of PCBs by lake trout (*Salvelinus namaycush*): an individual-based model approach. *Canadian Journal of Fisheries and Aquatic Sciences* 50: 97-109.

Magnan, P. 1988. Interactions between brook charr, *Salvelinus fontinalis*, and nonsalmonid species: ecological shift, morphological shift, and their impact on zooplankton communities. *Canadian Journal of Fisheries and Aquatic Sciences* 45: 999-1009.

Major, M.A., D.H. Rosenblatt, and K.A. Bostian. 1991. The octanol/water partition coefficient of methylmercury chloride and methylmercury hydroxide in pure water and salt solutions. *Environmental Toxicology and Chemistry* 10: 5-8.

Marais, J.F.K. and T. Erasmus. 1977. Chemical composition of the alimentary canal contents of mullet (Teleostei: Mugilidae) caught in the Swartkops Estuary near Port Elizabeth, South Africa. *Aquaculture* 10: 263-273.

- Marchettini, N., M. Panzneri, and E.B.P. Tiezzi. 2001. Effects of bioaccumulation of PCBs on biodiversity and distribution of fish in two creeks in east Tennessee (USA). *Ann. Chim.* 91: 435-443.
- Margenau, T.L., P.W. Rasmussen, and J.M. Kampa. 1998. Factors affecting growth of northern pike in small Wisconsin lakes. *North American Journal of Fisheries Management* 18: 625-639.
- Mason, A.Z. and K.D. Jenkins. 1995. Metal detoxification in aquatic organisms. *In Metal Speciation and Bioavailability in Aquatic Systems*. IUPAC Series on Analytical and Physical Chemistry of Environmental Systems. *Edited by* A. Tessier and D.R. Turner. John Wiley & Sons, pp. 479-608
- Matthiessen, P., G.F. Whale, R.J. Rycroft, and D.A. Sheahan. 1988. The joint toxicity of tank-mixes to rainbow trout. *Aquatic Toxicology* 13: 61-76.
- Mattingly, H.T. and M.J. Butler, IV. 1994. Laboratory predation on the Trinidadian guppy: implications for size-selective predation hypothesis and guppy life history evolution. *Oikos* 69: 54-64.
- McCarty, L.S. 1986. The relationship between aquatic toxicity QSARs and bioconcentration for some organic chemicals. *Environmental Toxicology and Chemistry* 5: 1071-1080.
- McCarty, L.S. and D. Mackay. 1993. Enhancing ecotoxicological modeling and assessment. *Environmental Science and Technology* 27: 1719-1728.
- McCarty, L.S., P.V. Hodson, G.C. Craig, and K.L.E. Kaiser. 1985. The use of quantitative structure-activity relationships to predict the acute and chronic toxicities of organic chemicals to fish. *Environmental Toxicology and Chemistry* 4: 595-606.
- McGurk, M.D. 1993. Allometry of herring mortality. *Transactions of the American Fisheries Society* 122: 1035-1042.
- McGurk, M.D. 1999. Size dependence of natural mortality rate of sockeye salmon and kokanee in freshwater. *North American Journal of Fisheries Management* 19: 376-396.
- McKim, J.M., J.W. Nichols, G.J. Lien, and S.L. Bertelsen. 1994. Respiratory-cardiovascular physiology and chloroethane gill flux in the channel catfish, *Ictalurus punctatus*. *Journal of Fish Biology* 44: 527-547.
- Meyer, H. 1899. Zur theorie der Alkoholnarkose, Erste Mittheilung. Welche Eigenschaft der Anesthetica bedingt ihre narkotische Wirkung? *Arch Exp Pathol Pharmacol* 42: 109-118.
- Middleton, A.J. and E.B. Smith. 1976. General anaesthetics and bacterial luminescence II. The effect of diethyl ether on the in vitro light emission of *Vibrio fischeri*. *Proceedings of the Royal Society of London Series B-biological Sciences* 193: 173-190.
- Miles, R.L. 1978. A life history study of the muskellunge in West Virginia. *American Fisheries Society Special Publication* 11: 140-145.
- Miller, K.W., W.B.M. Paton, R.A. Smith, and E.B. Smith. 1973. The pressure reversal of anesthesia and the critical volume hypothesis. *Molecular Pharmacology* 9: 131-143.
- Miller, T.J., L.B. Crowder, J.A. Rice, and F.P. Binkowski. 1992. Body size and the ontogeny of the functional response in fishes. *Canadian Journal of Fisheries and Aquatic Sciences* 49: 805-812.
- Miner, J.G. and R.A. Stein. 1996. Detection of predators and habitat choice by small bluegills: effects of turbidity and alternative prey. *Transactions of the American Fisheries Society* 125: 97-103.
- Minns, C.K. 1995. Allometry of home range size in lake and river fishes. *Canadian Journal of Fisheries and Aquatic Sciences* 52: 1499-1508.
- Minor, J.D. and E.J. Crossman. 1978. Home range and seasonal movements of muskellunge as determined by radiotelemetry. *American Fisheries Society Special Publication* 11: 146-153.
- Minton, J.W. and R.B. McLean. 1982. Measurements of growth and consumption of sauger (*Stizostedion canadense*): implications for fish energetics studies. *Canadian Journal of Fisheries and Aquatic Sciences* 39: 1396-1403.
- Mittelbach, G.G. and L. Persson. 1998. The ontogeny of piscivory and its ecological consequences. *Canadian Journal of Fisheries and Aquatic Sciences* 55: 1454-1465.
- Moharram, Y.G., S.A. El-Sharnouby, E.K. Moustaffa, and A. El-Soukary. 1987. Mercury and selenium in bouri (*Mugil cephalus*). *Water, Air, and Soil Pollution* 32: 455-459.
- Moore, C.M., R.J. Neves, J.J. Ney, and D.K. Whitehurst. 1985. Utilization of alewives and gizzard shad by striped bass in Smith Mountain Lake, Virginia. *Proceeding of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies* 39: 108-115.
- Morishita, T., U. Kazuaki, N. Imura, and T. Takahashi. 1987. Variation with growth in the proximate compositions of cultured red sea bream. *Nippon Suisan Gakkaishi* 53: 1601-1607.

- Mueller, G. 1996. Establishment of a fish community in the Hayden-Rhodes and Salt-Gila Aqueducts, Arizona. *North American Journal of Fisheries Management* 16: 795-804.
- Muir, B.S. and C.E. Brown. 1971. Effects of blood pathway on the blood-pressure drop in fish gills, with special reference to tunas. *Journal of the Fisheries Research Board of Canada* 28: 947-955.
- Muir, D.C.G., A.L. Yarechewski, D. A. Metner, and W.L. Lockhart. 1992. Dietary 2,3,7,8-tetrachlorodibenzofuran in rainbow trout: accumulation, disposition, and hepatic mixed-function oxidase enzyme induction. *Toxicology and Applied Pharmacology* 117: 65-74.
- Mullins, L.J. 1954. Some physical mechanisms in narcosis. *Chemical Reviews* 54: 289-323.
- Munn, M.D. and T.M. Short. 1997. Spatial heterogeneity of mercury bioaccumulation by walleye in Franklin D. Roosevelt Lake and the upper Columbia River, Washington. *Transactions of the American Fisheries Society* 126: 477-487.
- Murphy, G.W. 2004. Uptake of mercury and relationship to food habits of selected fish species in the Shenandoah River Basin, Virginia. Virginia Polytechnic Institute and State University, Blacksburg, VA.
- Myers, R.A. 1997. Comment and reanalysis: paradigms for recruitment studies. *Canadian Journal of Fisheries and Aquatic Sciences* 54: 978-981.
- Myers, R.A. and N.J. Barrowman. 1996. Is fish recruitment related to spawner abundance? *Fishery Bulletin* 94: 707-724.
- Neely, W.B. 1984. An analysis of aquatic toxicity data: water solubility and acute LC₅₀ fish data. *Chemosphere* 13: 813-819.
- Nichols, J.W., K.M. Jensen, J.E. Tietge, and R.D. Johnson. 1998. Physiologically based toxicokinetic model for maternal transfer of 2,3,7,8-tetrachlorodibenzo-p-dioxin in brook trout (*Salvelinus fontinalis*). *Environmental Toxicology and Chemistry* 17: 2422-2434.
- Niimi, A.J. and B.G. Oliver. 1987. Influence of molecular weight and molecular volume on the dietary absorption efficiency of chemicals by fish. *Canadian Journal of Fisheries and Aquatic Sciences* 45: 222-227.
- Norheim, G. and S.O. Roald. 1985. Distribution and elimination of hexachlorobenzene, octachlorostyrene, and decachlorobiphenyl in rainbow trout, *Salmo gairdneri*. *Aquatic Toxicology* 6: 13-24.
- Norstrom, R.J., A.E. McKinnon, and A.S.W. deFreitas. 1976. A bioenergetics-based model for pollutant accumulation by fish. Simulation of PCB and methylmercury residue levels in Ottawa River yellow perch (*Perca flavescens*). *Journal of the Fisheries Research Board of Canada* 33: 248-267.
- Norton, J.P. 1996. Roles of deterministic bounding in environmental modelling. *Ecological Modelling* 86: 157-161.
- Olem, H., S. Livengood, and K.M. Sandra. 1992. Lake and reservoir management. *Water Environmental Research* 64: 523-531.
- O'Loughlin, E.J., S.J. Traina, and Y.-P. Chin. 2000. Association of organotin compounds with aquatic and terrestrial humic substances. *Environmental Toxicology and Chemistry* 19: 2015-2021.
- Olson, R.J. and A.J. Mullen. 1986. Recent developments for making gastric evacuation and daily ration determinations. *Environmental Biology of Fishes* 16: 183-191.
- Opperhuizen, A. and S.M. Schrap. 1988. Uptake efficiencies of two polychlorobiphenyls in fish after dietary exposure to five different concentrations. *Chemosphere* 17: 253-262.
- Opstvedt, J., R. Miller, R.W. Hardy, and J. Spinelli. 1984. Heat-induced changes in sulfhydryl groups and disulfide bonds in fish protein and their effect on protein and amino acid digestibility in rainbow trout (*Salmo gairdneri*). *Journal of Agricultural and Food Chemistry* 32: 929-935.
- Overton, E. 1901. Studien über die Narkose, zugleich ein Beitrag zur allgemeiner Pharmakologie. Gustav Fischer Jena, Germany 00: 000-000.
- Paloheimo, J.E. and L.M. Dickie. 1965. Food and growth of fishes. I. A growth curve derived from experimental data. *Journal of the Fisheries Research Board of Canada* 22: 521-542.
- Park, R.A. and J.S. Clough. 2004. AQUATOX (Release 2) Modeling Environmental Fate and Ecological Effects in Aquatic Ecosystems volume 2: Technical Documentation. U.S. Environmental Protection Agency, Office of Water, Office of Technology and Science, Washington, DC. EPA/823/R-04/002.
- Parker, R.R. and P.A. Larkin. 1959. A concept of growth in fishes. *Journal of the Fisheries Research Board of Canada* 16: 721-745.

- Parks, J.W., C. Curry, D. Romani, and D.D. Russell. 1991. Young northern pike, yellow perch and crayfish as bioindicators in a mercury contaminated watercourse. *Environmental Monitoring and Assessment* 16: 39-73.
- Parsons, J.W. 1971. Selective food preferences of walleyes of the 1959 year class in Lake Erie. *Transactions of the American Fisheries Society* 100: 474-485.
- Paulson, L.J. 1980. Models of ammonia excretion for brook trout (*Salvelinus fontinalis*) and rainbow trout (*Salmo gairdneri*). *Canadian Journal of Fisheries and Aquatic Sciences* 37: 1421-1425.
- Persson, A. and L.-A. Hansson. 1999. Diet shift in fish following competitive release. *Canadian Journal of Fisheries and Aquatic Sciences* 56: 70-78.
- Peters, R.H. and J.V. Raelson. 1984. Relations between individual size and mammalian population density. *American Naturalist* 124: 498-517.
- Petersen, J.H. and D.L. Ward. 1999. Development and corroboration of a bioenergetics model for northern pikeminnow feeding on juvenile salmonids in the Columbia River. *Transactions of the American Fisheries Society* 128: 784-801.
- Peterson, D.R. 1994. Calculating the aquatic toxicity of hydrocarbon mixtures. *Chemosphere* 29: 2493-2506.
- Peterson, I. and J.S. Wroblewski. 1984. Mortality rates of fishes in the pelagic ecosystem. *Canadian Journal of Fisheries and Aquatic Sciences* 41: 1117-1120.
- Pierce, C.L., M.D. Sexton, M.E. Pelham, and H. Liao. 2001. Dynamics of the littoral fish assemblage in Spirit Lake, Iowa, and implications for prey availability for piscivores. *North American Journal of Fisheries Management* 21: 884-896.
- Piiper, J., P. Scheid, S.F. Perry, and G.M. Hughes. 1986. Effective and morphological oxygen-diffusing capacity of the gills of the elasmobranch *Scyliorhinus stellaris*. *Journal of Experimental Biology* 123: 27-41.
- Plante, C. and J.A. Downing. 1989. Production of freshwater invertebrate populations in lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 46: 1489-1498.
- Platt, S. and W.J. Hauser. 1978. Optimum temperature for feeding and growth of *Tilapia zillii*. *The Progressive Fish-Culturist* 40: 105-107.
- Playle, R.C., D.G. Dixon, and K. Burnison. 1993. Copper and cadmium binding to fish gills: estimates of metal-gill stability constants and modelling of metal accumulation. *Canadian Journal of Fisheries and Aquatic Sciences* 50: 2678-2687.
- Post, J.R., E.A. Parkinson, and N.T. Johnston. 1999. Density-dependent processes in structured fish populations: interaction strengths in whole-lake experiments. *Ecological Monographs* 69: 155-175.
- Press, W.H., S.A. Teukolsky, W.T. Vetterling, and B.P. Flannery. 1992. *Numerical Recipes in FORTRAN*. Cambridge University Press, Cambridge.
- Pringle, M.J., K.B. Brown, and K.W. Miller. 1981. Can the lipid theories of anesthesia account for the cutoff in anesthetic potency in homologous series of alcohols. *Molecular Pharmacology* 19: 49-55.
- Quinn, S.P. 1988. Flathead catfish abundance and growth in the Flint River, Georgia. *Proceedings of the Annual Conference of Southeastern Association of Fish and Wildlife Agencies* 42: 141-148.
- Radwell, A. 2000. Ecological integrity assessment of Ozark rivers to determine suitability for protective status, pp. 107. University of Arkansas, Fayetteville, AR.
- Railsback, S.F., B.C. Harvey, R.H. Lamberson, D.E. Lee, N.J. Claasen, and S. Yoshihara. 2002. Population-level analysis and validation of an individual-based cutthroat trout model. *Natural Resource Modeling* 15: 83-110.
- Ram, R.N. and J.W. Gillett. 1993. Comparison of alternative models for predicting the uptake of chlorinated hydrocarbons by oligochaetes. *Ecotoxicology and Environmental Safety* 26: 166-180.
- Rand, P.S., D.J. Stewart, P.W. Seelbach, M.L. Jones, and L.R. Wedge. 1993. Modeling steelhead population energetics in Lakes Michigan and Ontario. *Transactions of the American Fisheries Society* 122: 977-1001.
- Randall, R.G., J.R.M. Kelso, and C.K. Minns. 1995. Fish production in freshwaters: Are rivers more productive than lakes? *Canadian Journal of Fisheries and Aquatic Sciences* 52: 631-643.
- Rashevsky, N. 1959. Some remarks on the mathematical theory of nutrition of fishes. *Bulletin of Mathematical Biophysics* 21: 161-183.

- Rayner, J.M.V. 1985. Linear relations in biomechanics: the statistics of scaling functions. *Journal of Zoology*, London 206: 415-439.
- Reed, M.S. and C.F. Rabeni. 1989. Characteristics of an unexploited smallmouth bass population in a Missouri Ozark stream. *North American Journal of Fisheries Management* 9: 420-426.
- Reid, S.M., M.G. Fox, and T.H. Whillans. 1999. Influence of turbidity on piscivory in largemouth bass (*Micropterus salmoides*). *Canadian Journal of Fisheries and Aquatic Sciences* 56: 1362-1369.
- Reinert, K.H., J.M. Giddings, and L. Judd. 2002. Effects analysis of time-varying or repeated exposures in aquatic ecological risk assessment of argochemicals. *Environmental Toxicology and Chemistry* 21: 1977-1992.
- Richards, C.D., K. Martin, S. Gregory, C.A. Keightley, T.R. Hesketh, G.A. Smith, G.B. Warren, and J.C. Metcalfe. 1978. Degenerate perturbations of protein structure as the mechanism of anaesthetic action. *Nature (London)* 276: 775-779.
- Richards, R.J. 1959. A flexible growth function for empirical use. *Journal of Experimental Botany* 10: 290-300.
- Ricker, W.E. 1979. Growth rates and models *In* *Fish Physiology*. Volume VIII. *Edited by* W.S. Hoar, D.J. Randall and J.R. Brett. Academic Press, pp. 677-743
- Robinson, J.G. and K.H. Redford. 1986. Body size, diet and population density of neotropical forest mammals. *American Naturalist* 128: 665-680.
- Roch, M., J.A. McCarter, A.T. Matheson, M.J.R. Clark, and R.W. Olafson. 1982. Hepatic metallothionein in rainbow trout (*Salmo gairdneri*) as an indicator of metal pollution in the Campbell River system. *Canadian Journal of Fisheries and Aquatic Sciences* 39: 1596-1601.
- Roell, M.J. and D.J. Orth. 1993. Trophic basis of production of stream-dwelling smallmouth bass, rock bass, and flathead catfish in relation to invertebrate bait harvest. *Transactions of the American Fisheries Society* 122: 46-62.
- Roesijadi, G. 1992. Metallothionein in metal regulation and toxicity in aquatic animals. *Aquatic Toxicology* 22: 81-114.
- Roex, E.W.M., C.A.M. van Gestel, A.P. van Wezel, and N.M. van Straalen. 2000. Ratios between acute toxicity and effects on population growth rates in relation to toxicant mode of action. *Environmental Toxicology and Chemistry* 19: 685-693.
- Rose, K.A., E.S. Rutherford, D.S. McDermot, J.L. Forney, and E.L. Mills. 1999. Individual-based model of yellow perch and walleye populations in Oneida Lake. *Ecological Monographs* 69: 127-154.
- Ruohonen, K., D.J. Grove, and J.T. McIlroy. 1997. The amount of food ingested in a single meal by rainbow trout offered chopped herring, dry and wet diets. *Journal of Fish Biology* 51: 93-105.
- Saunders, R.L. 1962. The irrigation of the gills in fishes. II. Efficiency of oxygen uptake in relation to respiratory flow activity and concentrations of oxygen and carbon dioxide. *Canadian Journal of Zoology* 40: 817-862.
- Savitz, J. 1969. Effects of temperature and body weight on endogenous nitrogen excretion in the bluegill sunfish (*Lepomis macrochirus*). *Journal of the Fisheries Research Board of Canada* 26: 1813-1821.
- Schaeffer, J.S., R.C. Haas, J.S. Diana, and J.E. Breck. 1999. Field test of two energetic models for yellow perch. *Transactions of the American Fisheries Society* 128: 414-435.
- Schnute, J. 1981. A versatile growth model with statistically stable parameters. *Canadian Journal of Fisheries and Aquatic Sciences* 38: 1128-1140.
- Schreckenbach, K., R. Knösche, and K. Ebert. 2001. Nutrient and energy content of freshwater fishes. *Journal of Applied Ichthyology* 17: 142-144.
- Scott, D.P. and F.A.J. Armstrong. 1972. Mercury concentration in relation to size in several species of freshwater fish from Manitoba and northwestern Ontario. *Journal of the Fisheries Research Board of Canada* 29: 1685-1690.
- Selong, J.H., T.E. McMahon, A.V. Zale, and F.T. Barrows. 2001. Effect of temperature on growth and survival of bull trout, with application of an improved method for determining thermal tolerances in fishes. *Transactions of the American Fisheries Society* 130: 1026-1037.
- Shubina, L.I. and T.L. Rychagova. 1981. Dynamics of fat and water metabolism in Caspian shad, *Alosa caspia caspia*, in relation to its biological peculiarities (English Translation). *Journal of Ichthyology* 21(6): 123-127.
- Shultz, C.D., D. Crear, J.E. Pearson, J.B. Rivers, and J.W. Hylin. 1976. Total and organic mercury in the Pacific blue marlin.

- Bulletin of Environmental Contamination and Toxicology 15: 230-234.
- Sijm, D.T.H.M. and A. van der Linde. 1995. Size-dependent bioconcentration kinetics of hydrophobic organic chemicals in fish based on diffusive mass transfer and allometric relationships. *Environmental Science and Technology* 29: 2769-2777.
- Sijm, D.T.H.M., P. Pärt, and A. Opperhuizen. 1993. The influence of temperature on the uptake rate constants of hydrophobic compounds determined by the isolated perfused gills of rainbow trout (*Oncorhynchus mykiss*). *Aquatic Toxicology* 25: 1-14.
- Sijm, D.T.H.M., M.E. Verberne, P. Pärt, and A. Opperhuizen. 1994. Experimentally determined blood and water flow limitations for uptake of hydrophobic compounds using perfused gills of rainbow trout (*Oncorhynchus mykiss*): allometric applications. *Aquatic Toxicology* 30: 325-341.
- Sijm, D.T.H.M., M.E. Verberne, W.J. de Jonge, P. Pärt, and A. Opperhuizen. 1995. Allometry in the uptake of hydrophobic chemicals determined by in vivo and in isolated perfused gills. *Toxicology and Applied Pharmacology* 131: 130-135.
- Simkiss, K. 1983. Lipid solubility of heavy metals in saline solutions. *Journal of the Marine Biological Association of the United Kingdom* 63: 1-7.
- Simon, T.W. 1999. Two-dimensional Monte Carlo simulation and beyond: A comparison of several probabilistic Risk Assessment methods applied to a Superfund Site. *Human and Ecological Risk Assessment* 5: 823-843.
- Simpkins, D.G., W.A. Hubert, C. Martinez del Rio, and D.C. Rule. 2003. Physiological responses of juvenile rainbow trout to fasting and swimming activity: Effects on body composition and condition indices. *Transactions of the American Fisheries Society* 132: 576-589.
- Sompongse, W., Y. Itoh, and A. Obatake. 1996. Effect of cryoprotectants and a reducing reagent on the stability of actomyosin during ice storage. *Fisheries Science* 62: 73-79.
- Sood, C. and R.M. Bhagat. 2005. Interfacing geographical information systems and pesticide models. *Current Science* 89: 1362-1370.
- Sprenger, M.D., A.W. McIntosh, and S. Hoenig. 1988. Concentrations of trace elements in the yellow perch (*Perca flavescens*) from six acidic lakes. *Water, Air, and Soil Pollution* 37: 375-388.
- Stafford, C.P. and T.A. Haines. 1997. Mercury concentrations in Maine sport fishes. *Transactions of the American Fisheries Society* 126: 144-152.
- Staples, D.J. and M. Nomura. 1976. Influence of body size and food ration on the energy budget of rainbow trout *Salmo gairdneri* Richardson. *Journal of Fish Biology* 9: 29-43.
- Steen, J.B. and T. Berg. 1966. The gills of two species of haemoglobin-free fishes compared to those of other teleosts – with a note on severe anaemia in an eel. *Comparative Biochemistry and Physiology* 18: 517-526.
- Steingrímsson, S.O. and J.W.A. Grant. 1999. Allometry of territory size and metabolic rate as predictors of self thinning in young-of-year Atlantic salmon. *Journal of Animal Ecology* 68: 17-86.
- Stevens, E.D. and E.N. Lightfoot. 1986. Hydrodynamics of water flow in front of and through the gills of skipjack tuna. *Comparative Biochemistry and Physiology A* 83: 255-259.
- Stewart, D.J. and F.P. Binkowski. 1986. Dynamics of consumption and food conversion by Lake Michigan alewives: an energetics-modeling synthesis. *Transactions of the American Fisheries Society* 115: 643-659.
- Stewart, D.J. and M. Ibarra. 1991. Predation and production by salmonine fishes in Lake Michigan, 1978-88. *Canadian Journal of Fisheries and Aquatic Sciences* 48: 909-922.
- Stewart, D.J., D. Weininger, D.V. Rottiers, and T.A. Edsall. 1983. An energetics model for lake trout *Salvelinus namaycush*: application to the Lake Michigan population. *Canadian Journal of Fisheries and Aquatic Sciences* 40: 681-698.
- Stiefvater, R.J. and S.P. Malvestuto. 1985. Seasonal analysis of predator-prey size relationships in West Point Lake, Alabama-Georgia. *Proceeding of the Annual Conference of the Southeastern Association of Fish and Wildlife Agencies* 39: 19-27.
- Stober, J., D. Scheidt, R. Jones, K. Thornton, L. Gandy, J. Trexler, and S. Rathbun. 1998. South Florida Ecosystem Assessment Vol.1 Final Technical Report Phase 1. U.S. Environmental Protection Agency, Region IV and Office of Research and Development. EPA-904-R-98-002.
- Stockwell, J.D. and O.E. Johannsson. 1997. Temperature-dependent allometric models to estimate zooplankton production in temperate freshwater lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 54: 2350-2360.

- Storck, T.W. 1986. Importance of gizzard shad in the diet of largemouth bass in Lake Shelbyville, Illinois. *Transactions of the American Fisheries Society* 115: 21-27.
- Stow, C.A. and S.R. Carpenter. 1994. PCB accumulation in Lake Michigan coho and chinook salmon: individual-based models using allometric relationships. *Environmental Science and Technology* 28: 1543-1549.
- Summers, J.K., H.T. Wilson, and J. Kou. 1993. A method for quantifying the prediction uncertainties associated with water quality models. *Ecological Modelling* 65: 161-176.
- Swartzman, G.L. and R. Bentley. 1979. A review and comparison of plankton simulation models. *ISEM Journal* 1: 30-81.
- Sweka, J.A. and K.J. Hartman. 2001. Effects of turbidity on prey consumption and growth in brook trout and implications for bioenergetics modeling. *Canadian Journal of Fisheries and Aquatic Sciences* 58: 386-393.
- Sweka, J.A. and K.J. Hartman. 2003. Reduction of reactive distance and foraging success in smallmouth bass, *Micropterus dolomieu*, exposed to elevated turbidity levels. *Environmental Biology of Fishes* 67: 341-347.
- Takashi, R. 1973. Studies on muscular proteins of fish -VIII. Comparative studies on the biochemical properties of highly purified myosins from fish dorsal and rabbit skeletal muscle. *Bulletin of the Japanese Society of Scientific Fisheries* 39: 197-205.
- Takeda, H. and C. Shimizu. 1982. Purification of metallothionein from the liver of skipjack and its properties. *Bulletin of the Japanese Society of Scientific Fisheries* 48: 717-723.
- Tandler, A. and F.W.H. Beamish. 1981. Apparent specific dynamic action (SDA), fish weight and level of caloric intake in largemouth bass, *Micropterus salmoides* Lacepede. *Aquaculture* 23: 231-242.
- Tas, J.W., W. Seinen, and A. Opperhuizen. 1991. Lethal body burden of triphenyltin chloride in fish: preliminary results. *Comparative Biochemistry and Physiology C* 100: 59-60.
- Thomann, R.V. 1981. Equilibrium model of fate of microcontaminants in diverse aquatic food chains. *Canadian Journal of Fisheries and Aquatic Sciences* 38: 280-296.
- Thomann, R.V. 1989. Bioaccumulation model of organic chemical distribution in aquatic food chains. *Environmental Science and Technology* 23: 699-707.
- Thomann, R.V. and J.P. Connolly. 1984. Model of PCB in the Lake Michigan lake trout food chain. *Environmental Science and Technology* 18: 65-71.
- Thomann, R.V., J.P. Connolly, and T.F. Parkerton. 1992. An equilibrium model of organic chemical accumulation in aquatic food webs with sediment interaction. *Environmental Toxicology and Chemistry* 11: 615-629.
- Thornton, K.W. and A.S. Lessem. 1978. A temperature algorithm for modifying biological rates. *Transactions of the American Fisheries Society* 107: 284-287.
- Timmons, T.J., W.L. Shelton, and W.D. Davies. 1980. Differential growth of largemouth bass in West Point Reservoir, Alabama-Georgia. *Transactions of the American Fisheries Society* 109: 176-186.
- Tracey, D.M. 1993. Mercury levels in black cardinalfish (*Epigonus telescopus*). *New Zealand Journal of Marine and Freshwater Research* 27: 177-181.
- Turner, A.M., J.C. Trexler, C.F. Jordan, S.J. Slack, P. Geddes, J.H. Chick, and W.F. Loftus. 1999. Targeting ecosystem features for conservation: Standing crops in the Florida Everglades. *Conservation Biology* 13: 898-911.
- Ueda, T. and M. Takeda. 1983. Mercury and selenium levels in two species of sharks (In Japanese). *Bulletin of the Japanese Society of Scientific Fisheries* 49: 1731-1735.
- Umezawa, S.-I. and H. Watanabe. 1973. On the respiration of the killifish *Oryzias latipes*. *Journal of Experimental Biology* 58: 305-325.
- USEPA. 1994. EPA Superfund Record of Decision: Sangamo Weston/Twelve-mile Creek/Lake Hartwell PCB Contamination Superfund Site - Operable Unit Two Pickens, Pickens County, South Carolina (EPA ID: SCD003354412). U.S. Environmental Protection Agency, Region 4, Atlanta, GA. EPA/ROD/R04-94/178.
- USEPA. 2003. Methodology for Deriving Ambient Water Quality Criteria for the Protection of Human Health (2000) Technical Support Document Volume 2: Development of National Bioaccumulation Factors. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, Washington, DC. EPA/822/R-03/030.
- USEPA. 2004. Five-year Review Report for the Sangamo

- Weston/Twelvemile Creek/Lake Hartwell PCB Contamination Superfund Site - Operable Unit Two Pickens, Pickens County, South Carolina (EPA ID: SCD003354412) (<http://www.epa.gov/region04/waste/npl/nplsc/sangamo5yr.pdf>). U.S. Environmental Protection Agency, Region 4, Atlanta, GA.
- USEPA. 2005. Regulatory Impact Analysis for the Clean Air Mercury Rule. U.S. Environmental Protection Agency, Office of Air Quality Planning and Standards. EPA/452/R-05/003.
- van Hoogen, G. and A. Opperhuizen. 1988. Toxicokinetics of chlorobenzenes in fish. *Environmental Toxicology and Chemistry* 7: 213-219.
- van Loon, W.M.G.M., M.E. Verwoerd, F.G. Wijnker, C.J. van Leeuwen, P. van Duyn, C. van de Guchte, and J.L.M. Hermens. 1997. Estimating total body residues and baseline toxicity of complex organic mixtures in effluents and surface waters. *Environmental Toxicology and Chemistry* 16: 1358-1365.
- van Nes, E.H., E.H.R.R. Lammens, and M. Scheffer. 2002. PISCATOR, an individual-based model to analyze the dynamics of lake fish communities. *Ecological Modelling* 152: 261-278.
- van Veld, P.A., M.E. Bender, and M.H. Roberts, Jr. 1984. Uptake, distribution, and clearance of chlordecone by channel catfish (*Ictalurus punctatus*). *Aquatic Toxicology* 5: 33-49.
- van Wezel, A.P., D.T.H.M. Sijm, W. Seinen, and A. Opperhuizen. 1995. Use of lethal body burdens to indicate species differences in susceptibility to narcotic toxicants. *Chemosphere* 31: 3201-3209.
- Vandenbyllaardt, L., F.J. Ward, C.R. Braekevelt, and D.B. McIntyre. 1991. Relationships between turbidity, piscivory, and development of the retina in juvenile walleyes. *Transactions of the American Fisheries Society* 120: 382-390.
- VDEQ. 2005. PCB Strategy For the Commonwealth of Virginia. Virginia Department of Environmental Quality. <http://www.deq.virginia.gov/info/documents/PCB-Statewide-Strategy-2005.pdf>.
- Veith, G.D., D.J. Call, and L.T. Brooke. 1983. Structure-toxicity relationships for the fathead minnow, *Pimephales promelas*: narcotic industrial chemicals. *Canadian Journal of Fisheries and Aquatic Sciences* 40: 743-748.
- Verhaar, H.J.M., F.J.M. Busser, and J.L.M. Hermans. 1995. Surrogate parameter for the baseline toxicity content of contaminated water: simulating the bioconcentration of mixtures of pollutants and counting molecules. *Environmental Science and Technology* 29: 726-734.
- Vetter, R.D., M.C. Carey, and J.S. Patton. 1985. Coassimilation of dietary fat and benzo(a)pyrene in the small intestine: an absorption model using the killifish. *Journal of Lipid Research* 26: 428-434.
- Vogel, J.L. and D.A. Beauchamp. 1999. Effects of light, prey size, and turbidity on reaction distances of lake trout (*Salvelinus namaycush*) to salmonid prey. *Canadian Journal of Fisheries and Aquatic Sciences* 56: 1293-1297.
- Vorhees, D.J., S.B.K. Driscoll, K. von Stackelberg, and T.S. Bridges. 1998. Improving Dredged Material Management Decisions with Uncertainty Analysis. US Army Corps of Engineers Waterways Experiment Station, Washington, DC. Technical Report DOER-3.
- Walker, G. and T. Davies. 1974. Mass transfer in laminar flow between parallel permeable plates. *AIChE Journal* 20: 881-889.
- Wallach, D. and M. Genard. 1998. Effect of uncertainty in input and parameter values on model prediction error. *Ecological Modelling* 105: 337-345.
- Walter, J. 1973. Regular eigenvalue problems with eigenvalue parameter in the boundary condition. *Mathematische Zeitschrift* 133: 301-312.
- Wania, F. and D. Mackay. 1999. The evolution of mass balance models of persistent organic pollutant fate in the environment. *Environmental Pollution* 100: 223-240.
- Wanzenböck, J. and F. Schiemer. 1989. Prey detection in cyprinids during early development. *Canadian Journal of Fisheries and Aquatic Sciences* 46: 995-1001.
- Ware, D.M. 1975. Growth, metabolism, and optimal swimming speed of a pelagic fish. *Journal of the Fisheries Research Board of Canada* 32: 33-41.
- Waters, T.F., J.P. Kaehler, T.J. Polomis, and T.J. Kwak. 1993. Production dynamics of smallmouth bass in a small Minnesota stream. *Transactions of the American Fisheries Society* 122: 588-598.
- Watling, R.J., T.P. McClurg, and R.C. Stanton. 1981. Relation between mercury concentration and size in the mako shark. *Bulletin of Environmental Contamination and Toxicology* 26: 352-358.
- Weatherley, A.H. and H.S. Gill. 1983. Protein, lipid, water, and

caloric contents of immature rainbow trout, *Salmo gairdneri* Richardson, growing at different rates. *Journal of Fish Biology* 23: 653-673.

Weininger, D. 1978. Accumulation of PCBs by lake trout in Lake Michigan, pp. 232. University of Wisconsin, Madison, WI.

Werner, E.E. 1974. The fish size, prey size, handling time relation in several sunfishes and some implications. *Journal of the Fisheries Research Board of Canada* 31: 1531-1536.

Werner, E.E. and J.F. Gilliam. 1984. The ontogenetic niche and species interactions in size-structured populations. *Annual Review of Ecology and Systematics* 15: 393-425.

Wren, C.D. and H.R. MacCrimmon. 1986. Comparative bioaccumulation of mercury in two adjacent ecosystems. *Water Research* 20: 763-769.

Wright, P.A., D.J. Randall, and S.E. Perry III. 1989. Fish gill water boundary layer: a site of linkage between carbon dioxide and ammonia excretion. *Journal of Comparative Physiology* 158: 627-635.

Wu, L. and D.A. Culver. 1992. Ontogenetic diet shift in Lake Erie age-0 yellow perch (*Perca flavescens*): a size-related response to zooplankton density. *Canadian Journal of Fisheries and Aquatic Sciences* 49: 1932-1937.

Wurbs, R.A. 1994. Computer Models for Water Resources Planning and Management. U.S. Army Corps of Engineers, Institute for Water Resources, Alexandria, VA. IWR Report 94-NDS-7.

Yalkowsky, S.H., S.C. Valvani, and D. Mackay. 1983. Estimation of the aqueous solubility of some aromatic compounds. *Residue Reviews* 85: 43-55.

Yalkowsky, S.H., O.S. Carpenter, G.L. Flynn, and T.G. Slunick. 1973. Drug absorption kinetics in goldfish. *Journal of Pharmaceutical Sciences* 62: 1949-1954.

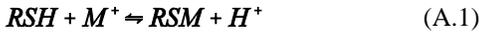
Yang, M.S. and P.A. Livingston. 1988. Food habits and daily ration of Greenland halibut, *Reinhardtius hippoglossoides*, in the eastern Bering Sea. *Fishery Bulletin* 86: 675-690.

APPENDICES

APPENDIX A. Equilibrium complexation model for metals

As reviewed by Mason and Jenkins (1995), metals can be classified into three different categories based on their complexation behavior and preference for different ligands. These groups are generally designated as class A, class B, and borderline metals. Of these, however, class B and borderline metals are the most important from an ecotoxicological point of view. Class B metals (e.g., Au, Ag, Cu, Hg, and Pb) preferentially bind to macromolecules such as proteins and nucleotides that are rich in sulfhydryl groups and heterocyclic nitrogen. Borderline metals (e.g., As, Cd, Co, Cr, Ni, Sn, and Zn) bind not only to the same sites as do class B metals but also to those sites preferred by class A metals (i.e., carboxylates, carbonyls, alcohols, phosphates, and phosphodiester). Although factors determining the preference of borderline metals for a particular binding site are complex, the fact that the transport and storage of these metals in fish and other biota are regulated by metallothioneins via sulfhydryl complexation reactions suggests that the total availability of sulfhydryl groups within organisms plays a key role in their internal distribution and accumulation.

To formulate complexation reactions for class B and borderline metals, one can assume that protein sulfhydryl groups are the only significant ligand for these metals, i.e.,



The stability constant for this reaction is

$$Kb = \frac{[RSM][H^+]}{[RSH][M^+]} = \frac{RSM[H^+]}{RSH[M^+]} \quad (A.2)$$

where $[H^+]$ is the hydrogen ion concentration (molar); $[M^+]$ is the concentration of free metal (molar); $[RSH]$ is the concentration of reactive sulfhydryls (molar); $[RSM]$ is the concentration of sulfur bound metal (molar); RSM is the moles of metal bound to sulfhydryls; and RSH is the moles of free, non-dissociated sulfhydryl. If a fish's metal concentrations (i.e., C_a , C_l , C_o , and C_p) are expressed on a molar basis, then the following identities hold

$$[M^+] = C_a \quad (A.3)$$

$$RSM = C_o P_o W_w \quad (A.4)$$

$$C_f = \left(P_a + P_l K_{ow} + P_o \frac{C_o}{C_a} \right) C_a \quad (A.5)$$

where W_w is the fish's kilogram live weight. Substituting Equations (A.3) and (A.4) into Equation (A.2), one can verify

that

$$\frac{P_o C_o}{C_a} = \frac{Kb RSH}{W_w [H^+]} \quad (A.6)$$

and consequently

$$C_f = \left(P_a + P_l K_{ow} + \frac{Kb RSH}{W_w [H^+]} \right) C_a \quad (A.7)$$

To parameterize Equation (A.7) for RSH , the following mass balance for the fish's sulfhydryl content is then assumed

$$\begin{aligned} TS &= RSH + RS^- + \sum_i RSM_i \\ &= RSH + \frac{RSH K_a}{[H^+]} + \sum_i \frac{Kb_i C_{a_i} RSH}{[H^+]} \\ &= RSH \left(1 + \frac{K_a}{[H^+]} + \sum_i \frac{Kb_i C_{a_i}}{[H^+]} \right) \end{aligned} \quad (A.8)$$

where TS is the total moles of sulfhydryl ligands; RS^- is the moles of disassociated sulfhydryls; and K_a is the sulfhydryl's disassociation constant. Therefore,

$$RSH = \frac{TS [H^+]}{[H^+] + K_a + \sum_i Kb_i C_{a_i}} \quad (A.9)$$

Using Equation (A.7), however, this expression can be rewritten as

$$RSH = \frac{TS}{1 + \frac{K_a}{[H^+]} + \sum_i \frac{Kb_i B_{f_i}}{(P_a + P_l K_{ow_i}) W_w [H^+] + Kb_i RSH}} \quad (A.10)$$

where $B_{f_i} = C_{f_i} W_w$ is the fish's total burden (mol/fish) of metal i . For most class B metals, however,

$$(P_a + P_l K_{ow_i}) W_w [H^+] \ll Kb_i RSH \quad (A.11)$$

Consequently, Equation (A.10) can be simplified to

$$RSH = \frac{TS}{1 + \frac{K_a}{[H^+]} + \sum_i \frac{B_{f_i}}{RSH}} = \frac{TS - \sum_i B_{f_i}}{1 + \frac{K_a}{[H^+]}} \quad (A.12)$$

This expression can then be substituted into Equation (A.7) to

calculate the fish aqueous phase metal concentrations.

To use the aforementioned complexation model (i.e., Equation (A.12) substituted into Equation (A.7)), one must specify both the metal's stability constant (see Equation (A.2)) and the total concentration of sulfhydryl binding sites TS (mol SH/g dry wt) within the fish. Although numerous studies have investigated the sulfhydryl content of selected fish tissues, it appears that no study has attempted to quantify the total sulfhydryl content of fish. A reasonable approximation of this parameter, however, can still be made since data do exist for the major tissues (i.e., muscle, liver, kidney, gill, and intestine) typically associated with metal bioaccumulation.

Itano and Sasaki (1983) reported the sulfhydryl content of Japanese sea bass (*Lateolabrax japonicus*) muscle to be 11.5 mol SH/g(sarcoplasmic protein) and 70.5 mol SH/g(myofibrillar protein). Using these authors' reported values of 0.0578 g(sarcoplasmic protein)/g(muscle) and 0.120 g(myofibrillar protein)/g(muscle), the total sulfhydryl content of Japanese sea bass muscle is estimated to be 9.12 mol(SH)/g(muscle) or 45.6 mol(SH)/g(dw muscle). Opstvedt et al. (1984) reported the sulfhydryl content of Pacific mackerel (*Pneumatophorus japonicus*) and Alaska pollock (*Theragra chalcogramma*) muscle to be 6.6 and 6.2 mmol(SH)/16 g(muscle N), respectively. Using conversion factors reported by these authors, these values are equivalent to 48.7 and 56.7 mol/g(dw muscle). Chung et al. (2000) determined the sulfhydryl content of mackerel (*Scomber australasicus*) muscle to be 88.2 mol(SH)/g(protein). Using the conversion factor 0.83 g(protein)/g(dw muscle) (Opstvedt et al. 1984), this value is equivalent to 73.2 mol(SH)/g(dw muscle). Several studies have determined sulfhydryl contents of the actomyosin and myosin components of fish myofibrillar proteins (Connell and Howgate 1959, Buttkeus 1967, 1971, Takashi 1973, Itoh et al. 1979, Sompongse et al. 1996, Benjakul et al. 1997, Lin and Park 1998). Because the results of these studies agree well with the actomyosin analysis reported by Itano and Sasaki (1983), the results of Itano and Sasaki (1983), Opstvedt et al. (1984), and Chung et al. (2000) can be assumed to be representative of fish in general. Consequently, the sulfhydryl content of fish muscle can be assumed to be on the order of 45-70 mol(SH)/g(dw muscle).

Although the sulfhydryl contents of liver, kidney, gills, and intestine have not been measured directly, the sulfhydryl content of these tissues can be estimated from their metallothionein concentrations. Metallothioneins (MT) are sulfur-rich proteins that are responsible for the transport and storage of heavy and trace metals and that are also usually considered to be the principal source of sulfhydryl binding sites in these tissues (Hamilton and Mehrle 1986, Roesijadi 1992). Numerous

researchers have investigated the occurrence of MTs in the liver, kidney, and gills of fish, and most have shown that tissue concentrations of MTs generally vary with metal exposures. Under moderate exposures, typical hepatic MT concentrations in fish are on the order of 0.03 - 0.30 mol(MT)/g(liver) (Brown and Parsons 1978, Roch et al. 1982, Klaverkamp and Duncan 1987, Dutton et al. 1993). Using data from Takeda and Shimizu (1982) who report the sulfhydryl content of skipjack tuna (*Katsuwonus pelamis*) MTs to be approximately 25 mol(SH)/mol(MT) and assuming a dry to wet weight ratio equal 0.2, these MT concentrations would be equivalent to 3.75 - 37.5 mol(SH)/g(dw liver). This range of values suggests that the hepatic sulfhydryl content of fish, that includes both baseline MT and cytoplasmic components that can be converted into MT, might be on the order of 40 mol(SH)/g(dw liver). This latter value, however, is probably too conservative. Consider, for example, the observation that the ratios of mercury concentrations in liver to those in muscle often vary from 1.5 to 6 or more (Lockhart et al. 1972, Shultz et al. 1976, Sprenger et al. 1988). If liver and muscle are equilibrating with the same internal aqueous phase, then either the MT sulfhydryls are more available than are the sarcoplasmic and myofibrillar sulfhydryls or the inducible concentrations of hepatic MT are much higher than 40 mol(SH)/g(dw liver). Of these two possibilities, the latter appears more likely.

Although gill, kidney, and intestine MTs have not been studied in the same detail as hepatic MTs, it appears that MT, and hence sulfhydryl, concentrations in gills and kidney are lower and not as inducible as hepatic concentrations (Hamilton et al. 1987a, b, Klaverkamp and Duncan 1987). Klaverkamp and Duncan (1987) estimated the concentrations of gill MT in white suckers (*Catostomus commersoni*) to be 33 g(MT)/g(gill) which is equivalent to 3.3 nmol(MT)/g(gill) or 0.0825 mol(SH)/g(gill). This latter value agrees well with the estimated concentrations of unidentified binding sites (0.03 - 0.06 mol/g(gill)) for copper on the gills of rainbow trout (*Oncorhynchus mykiss*) and brook trout (*Salvelinus fontinalis*) (MacRae et al. 1999), but is somewhat higher than the concentration of unidentified binding sites (0.013 - 0.03 mol/g(gill)) for copper, cadmium, and silver on the gills of rainbow trout and fathead minnows (*Pimephales promelas*) (Playle et al. 1993, Janes and Playle 1995).

Based on these considerations and the acknowledgment that many other important organic compounds contain sulfhydryl groups, e.g., enzymes involved in fatty acid synthesis, glutathione, etc., it seems reasonable to assume that the sulfhydryl content of fish is approximately 70 mol(SH)/g dry wt. Because Davis and Boyd (1978) reported the mean sulfur content of 17 fish species to be 206 mol(S)/g dry wt, this assumption implies that almost 1/3 of a fish's sulfur pool exists as sulfhydryl groups.

The aforementioned complexation model was implemented within BASS using 70 mol(SH)/g dry wt to calculate the fish's total sulfhydryl content. The mean dissociation constant for organic sulfhydryls was then assigned as $pK_a = 9.25$ (i.e., the SPARC estimated pK_a for cysteine). Using literature values for

the stability constants of methylmercury, however, BASS over predicted the bioaccumulation of methylmercury in fish by at least an order of magnitude. Consequently, a much simpler distribution coefficient algorithm was adopted.

APPENDIX B. Canonical equations for modeling diffusive chemical exchange across fish gills with ventilation and perfusion effects. See Section 2.2 for background information and notation.

If chemical exchange across fish gills is treated as steady-state, convective mass transport between parallel plates, then the following PDE and boundary conditions can be used to model chemical uptake from and excretion to the interlamellar water:

$$\frac{3}{2} \left(1 - \frac{x^2}{r^2} \right) V \frac{\partial C}{\partial y} = D \frac{\partial^2 C}{\partial x^2} \quad (\text{B.1})$$

$$\left. \frac{\partial C}{\partial x} \right|_{x=0} = 0 \quad (\text{B.2})$$

$$D \left. \frac{\partial C}{\partial x} \right|_{x=r} = -k_m \left[C(r, y) - C_a - \frac{2hD}{q_p} \int_y^l \left. \frac{\partial C}{\partial x} \right|_{x=r} dv \right] \quad (\text{B.3})$$

To obtain a canonical solution for this gill model, these equations can be nondimensionalized using the following transformations:

$$\Theta = \frac{C - C_a}{C_w - C_a} \quad (\text{B.4})$$

$$X = \frac{x}{r} \quad (\text{B.5})$$

$$Y = \frac{yD}{Vr^2} \quad (\text{B.6})$$

Applying these transformations, chemical exchange across a fish's gills is described by the following dimensionless PDE and boundary conditions:

$$\frac{3}{2} (1 - X^2) V \frac{\partial \Theta}{\partial Y} = \frac{\partial^2 \Theta}{\partial X^2} \quad (\text{B.7})$$

$$\left. \frac{\partial \Theta}{\partial X} \right|_{X=0} = 0 \quad (\text{B.8})$$

$$\left. \frac{\partial \Theta}{\partial X} \right|_{X=1} = -N_{Sh} \left[\Theta(1, Y) - \frac{2rhV}{q_p} \int_Y^{N_{Gz}} \left. \frac{\partial \Theta}{\partial X} \right|_{X=1} dv \right] \quad (\text{B.9})$$

where $N_{Sh} = k_m r D^{-1}$ is the gills' dimensionless lamellar permeability (i.e., Sherwood number); and $N_{Gz} = l D V^{-1} r^{-2}$ is the gills' dimensionless lamellar length (i.e., Graetz number).

The boundary condition (B.9) describing exchange across the secondary lamellae, however, can be simplified by noting that the

solution of Equation (B.7) is separable, i.e., $\Theta(X, Y) = \Phi(X)\Psi(Y)$ and that $q_v = 2rhV$ is the ventilation volume of an individual interlamellar channel. Using these observations, one can then write

$$\Psi(Y) \left. \frac{d\Phi}{dX} \right|_{X=1} = -N_{Sh} \left[\Phi(1)\Psi(Y) - \frac{q_v}{q_p} \left. \frac{d\Phi}{dX} \right|_{X=1} \int_Y^{N_{Gz}} \Psi(v) dv \right] \quad (\text{B.10})$$

that can then be differentiated with respect to Y to obtain

$$\frac{d\Psi}{dY} \left. \frac{d\Phi}{dX} \right|_{X=1} = -N_{Sh} \left[\Phi(1) \frac{d\Psi}{dY} + \Psi(Y) \frac{q_v}{q_p} \left. \frac{d\Phi}{dX} \right|_{X=1} \right] \quad (\text{B.11})$$

Because $\Psi(Y) = \exp(-\frac{2}{3}\lambda^2 Y)$ where λ is the constant of separation for Equation (B.7), the preceding equation is equivalent to

$$-\frac{2}{3}\lambda^2 \left. \frac{d\Phi}{dX} \right|_{X=1} = -N_{Sh} \left[-\frac{2}{3}\lambda^2 \Phi(1) + \frac{q_v}{q_p} \left. \frac{d\Phi}{dX} \right|_{X=1} \right] \quad (\text{B.12})$$

which can be rearranged to yield

$$\left. \frac{d\Phi}{dX} \right|_{X=1} = - \left(\frac{\frac{2}{3}\lambda^2 N_{Sh}}{\frac{2}{3}\lambda^2 - (q_v/q_p)N_{Sh}} \right) \Phi(1) \quad (\text{B.13})$$

Although this boundary condition is dependent on the eigenvalue λ , the eigenvalue expansion for the solution of Equation (B.7) is still straightforward (Walter 1973, Fulton 1977). Note that as the fish's perfusion rate increases, this boundary condition converges to

$$\left. \frac{d\Phi}{dX} \right|_{X=1} = -N_{Sh} \Phi(1) \quad (\text{B.14})$$

which is the boundary condition originally used by Barber et al. (1991).

See Barber et al. (1991) for the method used to construct the series solution for the dimensionless bulk concentration of the aforementioned PDE gill exchange model (i.e., Equation (2.28)).

APPENDIX C. Derivation of the consistency condition for feeding electivities.

To derive a self consistency condition for a fish's electivities and relative prey availabilities such that its calculated dietary frequencies will sum to unity, consider the following

$$e_i = \frac{d_i - f_i}{d_i + f_i} \quad (C.1)$$

$$e_i (d_i + f_i) = d_i - f_i \quad (C.2)$$

$$d_i = \left(\frac{1 + e_i}{1 - e_i} \right) f_i \quad (C.3)$$

Summing Equation (C.2) over all i then yields

$$\sum_{i=1}^n e_i (d_i + f_i) = \sum_{i=1}^n d_i - \sum_{i=1}^n f_i \quad (C.4)$$

$$\sum_{i=1}^n e_i (d_i + f_i) = 0 \quad (C.5)$$

When Equation (C.3) is substituted into Equation (C.5), one then obtains

$$\sum_{i=1}^n e_i \left(\frac{1 + e_i}{1 - e_i} f_i + f_i \right) = \sum_{i=1}^n \frac{2 e_i f_i}{1 - e_i} = 0 \quad (C.6)$$

or equivalently

$$\sum_{i=1}^n \frac{e_i f_i}{1 - e_i} = 0 \quad (C.7)$$

Finally, adding $\sum f_i = 1$ to each side of Equation (C.7), the following consistency condition is obtained

$$\sum_{i=1}^n \left(\frac{e_i f_i}{1 - e_i} + f_i \right) = 1 \quad (C.8)$$

$$\sum_{i=1}^n \frac{f_i}{1 - e_i} = 1$$

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