A. APPENDIX: PACIFIC SALMON POPULATION MODELING

Introduction

To assess the potential for adverse impacts of the pesticides bromoxynil and prometryn on Pacific salmon populations, a model was developed that explicitly links mortality due to exposure of young of the year to the productivity of salmon populations. We did this by constructing and analyzing general life-history matrix models for coho salmon (*Oncorhynchus kisutch*), sockeye salmon (*O. nerka*) and ocean-type and stream-type Chinook salmon (*O. tshawytscha*). The basic salmonid life history modeled consisted of hatching and rearing in freshwater, smoltification in estuaries, migration to the ocean, maturation at sea, and returning to the natal freshwater stream for spawning followed shortly by death. Differences between the modeled strategies are lifespan of the female, time to reproductive maturity, and the number and relative contribution of the reproductive age classes (Figure A1-1). The coho females we modeled reach reproductive maturity at age 3 and provide all of the reproductive contribution. Sockeye females reach maturity at age 4 or 5, but the majority of reproductive contributions are provided by age 4 females. Chinook females can mature at age 3, 4 or 5, with the majority of the reproductive contribution from ages 4 and 5. The primary difference between the ocean-type and stream-type Chinook is the juvenile freshwater residence with ocean-type juveniles migrating to the ocean as subyearlings and stream-type overwintering in freshwater and migrating to the ocean as yearlings. The models depicted general populations representing each life-history strategy and were constructed based upon literature data described below. Specific populations were not modeled due to the lack of sufficient demographic and reproductive data for a single population.

The acute toxicity model estimated the population-level impacts of juvenile mortality resulting from exposure to lethal concentrations of contaminants. These models excluded sublethal and indirect effects of the exposures and focused on the population-level outcomes resulting from an annual exposure of juveniles to a pesticide. The lethal impact was implemented as a change in first year survival for each of the salmon life-history strategies.

The overall model endpoint used to assess population-level impacts for the acute lethality model was the percent change in the intrinsic population growth rate (lambda, λ) resulting from the pesticide exposure. Change in λ is an accepted population parameter often used in evaluating population productivity, status, and viability. The National Marine Fisheries Service uses changes in λ when estimating the status of species, conducting risk and viability assessments, developing Endangered Species Recovery Plans, composing Biological Opinions, and communicating with other federal, state and local agencies (McClure et al. 2003). While values of $\lambda<1.0$ indicate a declining population, negative changes in lambda greater than the natural variability for the population indicate a loss of productivity. This can be a cause for concern since the decline could make a population more susceptible to dropping below 1.0 due to impacts from multiple stressors.

Assessing the results from different pesticide exposure scenarios relative to a control (i.e. unexposed) scenario can indicate the potential for pesticide exposures to lead to changes in the first year survival. Consequently, subsequent changes in salmon population dynamics as indicated by percent change in a population's intrinsic rate of increase assists in forecasting the potential population-level impacts to listed populations. The model conveys the potential influence of life-history strategies that might explain differential results within the species modeled.

Methods

In order to understand the relative impacts of a short-term pesticide exposure on exposed vs. unexposed fish, we used parameters for an idealized baseline population that exhibits an increasing population growth rate. All characteristics exhibit density independent dynamics. There were no definitive data available on the populations to support specific density dependent relationships, so rather than assign an unsupported relationship, the National Research Council recommendation was followed to utilize density independent parameters (NAS NRC 2013). The models assume closed systems, allowing no migration impact on population size. No stochastic impacts are included beyond natural variability reported in the literature as represented by selecting parameter values from a normal distribution about a mean each model iteration (year). Ocean conditions, freshwater habitat, fishing pressure, and marine resource availability were assumed constant and density independent so that they remain in the range they occupied during the period when demographic data were collected.

In the model an individual fish experiences an exposure scenario once as a subyearling (during its first spring) and never again. The pesticide exposure is assumed to occur to the population annually. All individuals in one cohort within a given population are assumed to be exposed to the pesticide during their subyearling spring-summer growth period. No other age classes experience the exposure.

A prospective analysis of the transition matrix, A, (Caswell 2001) explored the intrinsic population growth rate as a function of the vital rates. The intrinsic population growth rate, λ, equals the dominant eigenvalue of A and was calculated using matrix analysis software (MATLAB version 7.7.0 by The Math Works Inc., Natick, MA). Therefore λ is calculated directly from the matrix and running projections of abundances over time is redundant and unnecessary. The stable age distribution, the proportional distribution of individuals among the ages when the population is at equilibrium, is calculated as the right normalized eigenvector corresponding to the dominant eigenvalue λ. Variability was integrated by repeating the calculation of λ 2000 times selecting the values in the transition matrix from their normal distribution defined by the mean standard deviation. The influence of each matrix element, a_{ij}, on λ was assessed by calculating the sensitivity values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to a_{ij}, defined by $\delta\lambda / \delta a_{ij}$. Higher sensitivity values indicate greater influence on λ. The elasticity of matrix element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. One characteristic of elasticity analysis is that the elasticity values for a transition matrix sum to unity (one). The unity characteristic also allows comparison of the influence of transition elements and comparison across matrices.

Due to differences in the life-history strategies, specifically lifespan, age at reproduction and first year residence and migration habits, four life-history models were constructed. The differences in life history may result in different freshwater pesticide exposure profiles which can translate into potentially different population-level responses. Separate models were constructed for coho salmon, sockeye salmon, ocean-type and stream-type Chinook salmon. In all cases, transition values were determined from literature data on survival and reproductive characteristics of each species for populations that exhibit the life history strategy and were listed as endangered, threatened, or a species of concern under the ESA. All transition values are listed in Table A1-1.

A life-history transition matrix was constructed for coho salmon (*O. kisutch*) with a maximum age of 3. Spawning occurs in late fall and early winter with emergence from March to May. Fry spend 14-18 months in freshwater, smolt and spend 16-20 months in the saltwater before returning to spawn (Pess et al. 2002). Survival numbers were summarized in Knudsen et al. (2002) as follows. The average fecundity of each female is 4500 with a standard deviation of 500. The observed number of males:females was 1:1. Survival from spawning to emergence is 0.3 (0.07). Survival from emergence to smolt is 0.0296 (0.00029) and marine survival is 0.05 (0.01). All parameters followed a normal distribution (Knudson et al. 2002). The calculated values used in the matrix are listed in Table A1-1. The growth period for first year coho was set at 180 days to represent the time from mid-spring to mid-fall when the temperatures and resources drop and somatic growth slows (Knudson et al. 2002).

The life-history matrix for sockeye salmon (*O. nerka*) were based upon the lake wintering populations of Lake Washington, Washington, USA. These female sockeye salmon spend one winter in freshwater, then migrate to the ocean to spend three to four winters before returning to spawn at ages 4 or 5. Jacks return at age 2 after only one winter in the ocean. The age proportion of returning adults is 0.03, 0.82, and 0.15 for ages 3, 4 and 5, respectively (Gustafson et al.1997). All age 3 returning adults are males. Hatch rate and first year survival were calculated from brood year data on escapement, resulting presmolts and returning adults (Pauley et al. 1989) and fecundity (McGurk 2000). Fecundity values for age 4 females were 3374 (473) and for age 5 females were 4058 (557) (McGurk 2000). First year survival rates were 0.737/month (Gustafson et al. 1997). Ocean survival rates were calculated based upon brood data and the findings that 90% of ocean mortality occurs during the first 4 months of ocean residence (Pauley et al. 1989). Matrix values used in the sockeye baseline model are listed in Table A1-1. The 168 day growth period represents the time from lake entry to early fall when the temperature drops and somatic growth slows (Gustafson et al. 1997).

A life-history matrix was constructed for ocean-type Chinook salmon (*O. tshawytscha*) with a maximum female age of 5 and reproductive maturity at ages 3, 4 or 5. Ocean-type Chinook migrate from their natal stream within a couple months of hatching and spend several months rearing in estuary and nearshore habitats before continuing on to the open ocean. Transition values were determined from literature data on survival and reproductive characteristics from several ocean-type Chinook populations in the Columbia River system (Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Greene and Beechie 2004). The sex ratio of spawners was approximately 1:1. Estimated size-based fecundity of 4511(65), 5184(89), and 5812(102) was calculated based on data from Howell et al., 1985, using length-fecundity relationships from Healy and Heard (1984). Control matrix values for the

Chinook model are listed in Table A1-1. The growth period of 140 days encompasses the time the fish rear in freshwater prior to entering the estuary and open ocean. The first three months of estuary/ocean survival are the size-dependent stage. Size data for determining subyearling Chinook condition indices came from data collected in the lower Columbia River and estuary (Johnson et al. 2007).

An age-structured life-history matrix for stream-type Chinook salmon with a maximum age of 5 was defined based upon literature data on Yakima River spring Chinook from Knudsen et al. (2006) and Fast et al. (1988), with sex ratios of 0.035, 0.62 and 0.62 for females spawning at ages 3, 4, and 5, respectively. Length data from Fast et al. (1988) was used to calculate fecundity from the length-fecundity relationships in Healy and Heard (1984). The 184-day growth period produces control fish with a mean size of 96mm, within the observed range documented in the fall prior to the first winter (Beckman et al. 2000). The size-dependent survival encompasses the 4 early winter months, up until the fish are 12 months old.

Acute Toxicity Model

In order to estimate the population-level responses of exposure to lethal pesticide concentrations, acute mortality models were constructed based upon the control life-history matrices described above. The acute responses are modeled as direct reduction in the first year survival rate (S1). Two options are available to run, direct mortality estimates and exposure scenarios. Direct mortality can be input as percent mortality and is multiplied by the first-year survival rate in the transition matrix. Calculated EEC values can be assessed in the Risk-Plots to identify the appropriate level of mortality. In contrast, modelling exposure scenarios results in a cumulative reduction in survival as defined by the concentration and the dose-response curve (the LC_{50} and slope for each pesticide). A sigmoid dose-response relationship is used to accurately handle responses well away from LC_{50} and to be consistent with other does-response relationships. The model inputs for each scenario are the exposure concentration and acute fish LC_{50}, as well as the sigmoid slope for the LC_{50}. For a given concentration, a pesticide survival rate (1-mortality) is calculated and is multiplied by the control first-year survival rate, producing an exposed scenario first-year survival for the life-history matrix. The model allows for a specified percentage of the population (0-100%) to experience the exposure.

Demographic variability is incorporated as described above using mean and standard deviation of normally distributed survival and reproductive rates and model output consists of the percent change in lambda from unexposed control populations derived from the mean of 10000 calculations of both the unexposed control population and the pesticide exposed population. For the purposes of this assessment, the percent change in lambda is defined as different from control when the difference between the mean percent change is greater than the percent of one standard deviation from the control lambda.

For this exercise, maximum EEC values on the Risk-plots relate to juvenile mortality of 1% and 10%. We determined the population-level response by varying the percent of the population exposed (10, 50, and 100%).

Results

Sensitivity Analysis

The sensitivity analysis of all four of the control population matrices predicted the greatest changes in population growth rate (λ) result from changes in first-year survival. Parameter values and their corresponding sensitivity values are listed in Table A1-1. The elasticity values for the transition matrices also corresponded to the driving influence of first-year survival, with contributions to lambda of 0.33 for coho, 0.29 for ocean-type Chinook, 0.25 for stream-type Chinook, and 0.24 for sockeye.

Model Output

While trends in effects were seen for each pesticide across all four life-history strategies modeled, some slight differences were apparent. The similarity in patterns likely stems from using the same toxicity values for all four salmon, while the differences are consequences of distinctions between the life-history matrices. The stream-type Chinook and sockeye models produced very similar results as measured as the percent change in population growth rate. The ocean-type Chinook and coho models output produced the greatest changes in lambda resulting from the pesticide exposures. When looking for similarities in parameters to explain the ranking, no single life history parameter or characteristic, such as lifespan, reproductive ages, age distribution, lambda and standard deviation, or first-year survival show a pattern that matches this consistent output. Combining these factors into the transition matrix for each life-history and conducting the sensitivity and elasticity analyses revealed that changes in first-year survival produced the greatest changes in lambda. In addition, the elasticity analysis can be used to predict relative contribution to lambda from changes in first-year survival on a per unit basis. As detailed by the elasticity values reported above, the same change in first-year survival will produce a slightly greater change in the population growth rate for coho and ocean-type Chinook than for stream-type Chinook and sockeye. While some life-history characteristics may lead a population to be more vulnerable to an impact, the culmination of age structure, survival and reproductive rates as a whole strongly influences the population-level response.

Slight shifts in population growth rate occurred for the 10% mortality levels and increased with the percentage of the population exposed, but did not exceed the standard deviation for any of the life-history strategies (Tables A1-2-5). Model results for stream-type Chinook salmon approached significant impacts primarily due to the size of the standard deviation of the unexposed population. Percent changes in lambda were considered significant if they were outside of one standard deviation from the unexposed population. These results indicate that exposure of salmonid populations to bromoxynil and prometryn at the EECs calculated based upon the label application rates would not have consequences to the population's growth rate greater than the natural variability observed in the data used to parameterize the model.

When we compare the model output concentrations to calculated expected environmental concentrations in salmonid habitats described in the exposure section, it is likely that some individuals within a population will be exposed during their freshwater juvenile lifestage, particularly those juveniles exposed while utilizing off-channel habitats. The likelihood of population effects from death of juveniles increases for those populations that spend longer periods in freshwaters such as stream-type Chinook, sockeye, and coho salmon.

For those populations with lambdas greater than one, reductions in lambda from death of subyearlings can lead to consequences to abundance and productivity. Attainment of recovery and time-associated goals would be delayed for populations with reduced lambdas. For those natural populations with current lambdas of less than one, risk of extinction would increase. Many of the populations that are categorized as core populations or are important to individual strata, have lambdas just above one and are essential to survival and recovery goals. Slight changes in lambda, even as small as 3-4%, would result in reduced abundances and increased time to meet population recovery goals.

References

Beckman, B.R., Larsen, D.A., Sharpe, C., Lee-Pawlak, B., Schreck, C.B., and Dickhoff, W.W. 2000. Physiological status of naturally reared juvenile spring Chinook salmon in the Yakima River: seasonal dynamics and changes associated with smolting. Transactions of the American Fisheries Society 129:727–753.

Caswell, H. 2001. Matrix population models: Construction, analysis, and interpretation. Sundarland, MA, USA: Sinauer Assoc.

Fast, D.E., Hubble, J.D., and Kohn, M.S. 1988. Yakima River Spring Chinook Enhancement Study, Annual Report FY 1988.U.S. Department of Energy, Bonneville Power Administration Division of Fish and Wildlife. Project No. 82-16. 101pp.

Greene, C.M., and Beechie, T.J. 2004. Consequences of potential density-dependent mechanisms on recovery of ocean-type Chinook salmon (Oncorhynchus tshawytscha). Canadian Journal of Fisheries and Aquatic Sciences 61:590-602.

Gustafson, R.G., Wainwright, T.C., Winans, G.A., Waknitz, F.W., Parker, L.T., and Waples, R.S. 1997. Status review of sockeye salmon from Washington and Oregon. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-NWFSC-33, 282 pp.

Healey, M.C. 1991. Life history of Chinook salmon (Oncorhynchus tshawytscha). Pages 311–394 in C. Groot and L. Margolis, editors. Pacific salmon life histories. University of British Columbia Press, Vancouver, Canada.

Healey, M.C., and Heard, W.R. 1984. Inter- and intra-population variation in the fecundity of Chinook salmon (Oncoryhchus tshawytscha) and its relevance to life history theory. Canadian Journal of Fisheries and Aquatic Sciences 41:476-483.

Howell, P., Jones, K., Scarnecchia, D., LaVoy, L., Kendra, W., Ortmann, D., Neff, C., Petrosky, C., and Thurow, R. 1985. Stock assessment of Columbia River anadromous salmonids Volume I: Chinook, coho, chum, and sockeye salmon stock summaries. Final Report to Bonneville Power Administration. Bonneville Power Administration, P.O Box 3621, Portland OR 97208, DE-AI79-84BP12737, Project No. 83-335. 579 p.

Johnson, L.L., Ylitalo, G.M., Arkoosh, M.R., Kagley, A.N., Stafford, C.L., Bolton, J.L., Buzitis, J., Anulacion, B.F., and Collier, T.K. 2007. Contaminant exposure in outmigrant juvenile salmon from Pacific Northwest estuaries. Environmental Monitoring and Assessment 124:167-194.

Knudsen, C.M., Schroder, S.L., Busack, C.A., Johnston, M.V., Pearsons, T.N., Bosch, W.J., and Fast, D.E. 2006. Comparison of life history traits between first-generation hatchery and wild upper Yakima River spring Chinook salmon. Transactions of the American Fisheries Society 135:1130-1144.

Knudsen, E.E., Symmes, E.W., and Margraf, F.J. 2002. Searching for an ecological life history approach to salmon escapement management. American Fisheries Society Symposium 34:261-276.

McClure, M.M., Holmes, E.E., Sanderson, B.L., and Jordan, C.E. 2003. A large-scale, multispecies status assessment: anadromous salmonids in the Columbia River Basin. Ecological Applications 13(4):964-989.

McGurk, M.D. 2000. Comparison of fecundity–length–latitude relationships between nonanadromous (kokanee) and anadromous sockeye salmon (Oncorhynchus nerka). Canadian Journal of Zoology 78: 1791–1805.

NAS NRC 2013

PSCCTC (Pacific Salmon Commission Chinook Technical Committee). 2002. Pacific Salmon Commission Joint Chinook Technical Committee Report: Annual Exploitation Rate Analysis and Model Calibration. Report TCCHINOOK (02)-3. Vancouver, British Columbia, Canada.

Pauley, G.B., Risher, R., and Thomas, G.L. 1989. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest) - sockeye salmon. U.S. Fish Wildl. Serv., Biol. Rep. 82(11.116). U.S. Army Corps of Engineers, TR EL-82-4, 22 p.

Pess, G.R., Montgomery, D.R., Steel, E.A., Bilby, R.E., Feist, B.E., and Greenberg, H.M. 2002. Landscape characteristics, land use, and coho salmon (Oncorhynchus kisutch) abundance, Snohomish River, Wash., U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 59: 613–623.

Quinn, T.P. 2005. The behavior and ecology of Pacific salmon and trout. University of Washington Press. Seattle, WA. 378pp.

Ratner, S., Lande, R., and Roper, B.B. 1997. Population viability analysis of spring Chinook salmon in the South Umpqua River, Oregon. Conservation Biology 11:879-889.

Relyea, R.A. 2005. The impact of insecticides and herbicides on the biodiversity and productivity of aquatic communities. Ecological Applications 15(2): 618–627.

Roni, P., and Quinn, T.P. 1995. Geographic variation in size and age of North American Chinook salmon. North American Journal of Fisheries Management 15:325-345.

Table A1-1. Matrix transition element (standard deviation) and sensitivity (S) and elasticity (E) values for each model species. These control values are listed by the transition element taken from the life-history graphs as depicted in Figure A1-1 and the literature data described in the method text. Blank cells indicate elements that are not in the transition matrix for a particular species. The influence of each matrix element on λ was assessed by calculating the sensitivity (S) and elasticity (E) values for A. The sensitivity of matrix element a_{ij} equals the rate of change in λ with respect to the transition element, defined by $\delta\lambda / \delta a$. The elasticity of transition element a_{ij} is defined as the proportional change in λ relative to the proportional change in a_{ij}, and equals (a_{ij}/λ) times the sensitivity of a_{ij}. Elasticity values allow comparison of the influence of individual transition elements and comparison across matrices.

Transition Element	Chinook Stream-type			Chinook Ocean-type			Coho			Sockeye		
	Value[1] (std)	S	E	Value[2] (std)	S	E	Value[3] (std)	S	E	Value[4]	S	E
S1	0.0643 (0.003)	3.844	0.247	0.0056 (0.001)	57.13	0.292	0.0296 (0.002)	11.59	0.333	0.0257 (0.003)	9.441	0.239
S2	0.1160 (0.002)	2.132	0.247	0.48 (0.097)	0.670	0.292	0.0505 (0.005)	6.809	0.333	0.183 (0.003)	1.326	0.239
S3	0.17006 (0.004)	1.448	0.246	0.246 (0.050)	0.476	0.106				0.499 (0.003)	0.486	0.239
S4	0.04 (0.002)	0.319	0.0127	0.136 (0.023)	0.136	0.0168				0.1377 (0.003)	0.322	0.0437
R3	0.5807 (0.089)	0.00184	0.0011	313.8 (38.1)	0.0006	0.186	732.8 (75.0)	0.000469	0.333			
R4	746.73 (86.62)	0.000313	0.233	677.1 (80.7)	0.000146	0.0896				379.57 (53.2)	0.000537	0.195
R5	1020.36 (101.33)	1.25E-05	0.0127	1028 (117.5)	1.80E-05	0.0168				608.7 (83.0)	7.28E-05	0.0437

[1] Value calculated from data in Healey and Heard 1984, Fast et al. 1988, Beckman et al. 2000, Knudsen et al. 2006
[2] Value calculated from data in Healey and Heard 1984, Howell et al. 1985, Roni and Quinn 1995, Ratner et al. 1997, PSCCTC 2002, Green and Beechie 2004, Johnson et al. 2007
[3] Value calculated from data in Pess et al. 2002, Knudsen et al. 2002
[4] Value calculated from data in Pauley et al. 1989, Gustafson et al. 1997, McGurk 2000

Table A1-2. Acute mortality model output for ocean-type Chinook. The percent change in population growth rate (lambda), the population growth rate mean values (lambda mean), and the variability (standard deviations) are shown, along with the mean first year survival rate (S1). The toxicity values were applied as direct mortality on first year survival. The percent of the population exposed was also varied (left column). Bold indicates a percent change in population growth rage of greater than one standard deviation from control values (second column).

% of population exposed	Chinook, ocean-type		**% Mortality from exposure to EEC**	
	control		**10**	**1**
100		% change lambda	-3	0
	9	% change lambda std	12.4	12.8
	1.09	lambda mean	1.06	1.09
	0.1	lambda std	0.1	0.1
	5.64E-03	S1	5.08E-03	5.58E-03
50		% change lambda	-1	0
		% change lambda std	12.9	12.8
		lambda mean	1.08	1.09
		lambda std	0.1	0.1
		S1	5.36E-03	5.62E-03
10		% change lambda	0	0
		% change lambda std	12.9	13
		lambda mean	1.09	1.09
		lambda std	0.1	0.1
		S1	5.57E-03	5.62E-03

Table A1-3. Acute mortality model output for stream-type Chinook. The percent change in population growth rate (lambda), the population growth rate mean values (lambda mean), and the variability (standard deviations) are shown, along with the mean first year survival rate (S1). The toxicity values were applied as direct mortality on first year survival. The percent of the population exposed was also varied (left column). Bold indicates a percent change in population growth rage of greater than one standard deviation from control values (second column).

% of population exposed	Chinook, stream-type		**% Mortality from exposure to EEC**	
	control		**10**	**1**
100		% change lambda	-3	0
	3	% change lambda std	4.3	4.4
	1	lambda mean	0.97	1
	0.03	lambda std	0.03	0.03
	6.43E-02	S1	5.79E-02	6.37E-02
50		% change lambda	-1	0
		% change lambda std	4.5	4.4
		lambda mean	0.99	1
		lambda std	0.03	0.03
		S1	6.11E-02	6.40E-02
10		% change lambda	0	0
		% change lambda std	4.4	4.4
		lambda mean	1	1
		lambda std	0.03	0.03
		S1	6.37E-02	6.43E-02

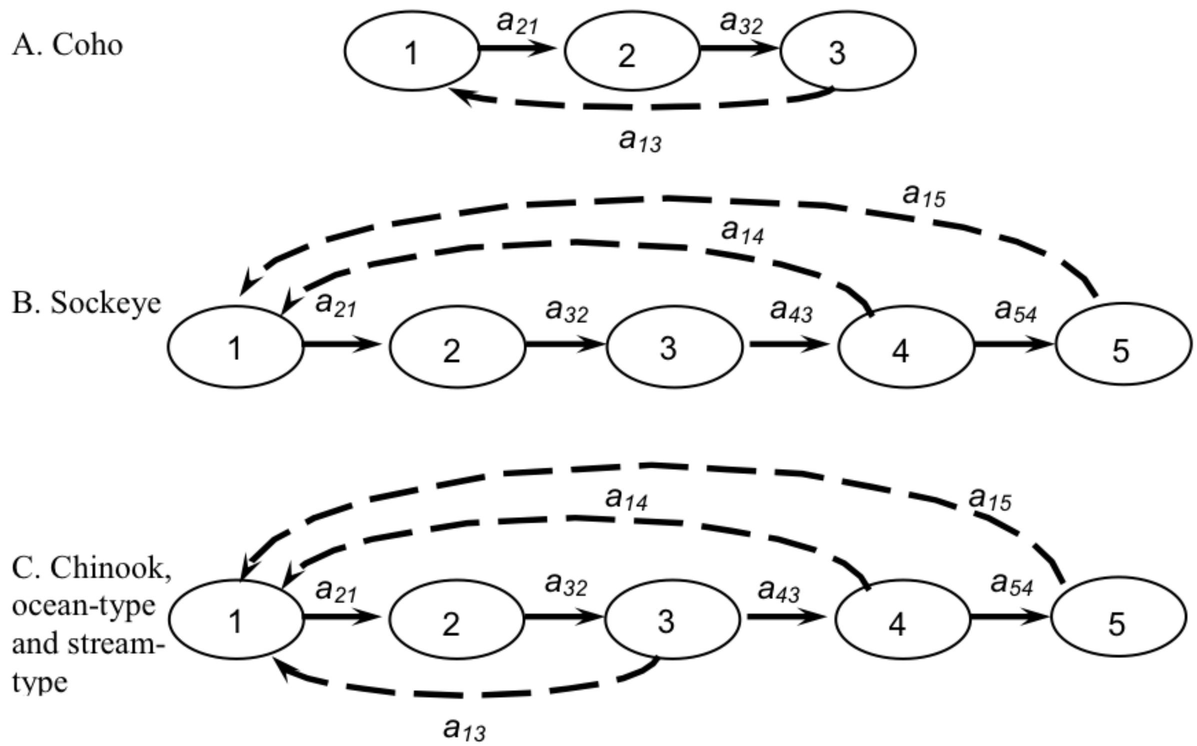
Table A1-4 Acute mortality model output for sockeye. The percent change in population growth rate (lambda), the population growth rate mean values (lambda mean), and the variability (standard deviations) are shown, along with the mean first year survival rate (S1). The toxicity values were applied as direct mortality on first year survival. The percent of the population exposed was also varied (left column). Bold indicates a percent change in population growth rage of greater than one standard deviation from control values (second column).

% of population exposed	Sockeye		**% Mortality from exposure to EEC**	
	control		**10**	**1**
100		% change lambda	-2	0
	6	% change lambda std	7.7	7.9
	1.01	lambda mean	0.99	1.01
	0.06	lambda std	0.05	0.06
	2.57E-02	S1	2.31E-02	2.54E-02
50		% change lambda	-1	0
		% change lambda std	8	8
		lambda mean	1	1.01
		lambda std	0.06	0.06
		S1	2.44E-02	2.56E-02
10		% change lambda	0	0
		% change lambda std	7.9	7.9
		lambda mean	1.01	1.01
		lambda std	0.06	0.06
		S1	2.54E-02	2.57E-02

Table A1-5. Acute mortality model output for coho. The percent change in population growth rate (lambda), the population growth rate mean values (lambda mean), and the variability (standard deviations) are shown, along with the mean first year survival rate (S1). The toxicity values were applied as direct mortality on first year survival. The percent of the population exposed was also varied (left column). Bold indicates a percent change in population growth rage of greater than one standard deviation from control values (second column).

% of population exposed	Coho		**% Mortality from exposure to EEC**	
	control		**10**	**1**
100		% change lambda	-4	0
	5	% change lambda std	7.2	7.4
	1.03	lambda mean	0.99	1.03
	0.05	lambda std	0.05	0.05
	2.97E-02	S1	2.67E-02	2.94E-02
50		% change lambda	-2	0
		% change lambda std	7.5	7.4
		lambda mean	1.01	1.03
		lambda std	0.06	0.05
		S1	2.81E-02	2.95E-02
10		% change lambda	0	0
		% change lambda std	7.5	7.4
		lambda mean	1.02	1.03
		lambda std	0.06	0.05
		S1	2.94E-02	2.96E-02

Figure A1-1: Life-History Graphs and Transition Matrix for coho (A), sockeye (B) and Chinook (C) salmon. The life-history graph for a population labeled by age, with each transition element labeled according to the matrix position, a_{ij}, i row and j column. Dashed lines represent reproductive coCntribution and solid lines represent survival transitions. D) The transition matrix for the life-history graph depicted in C.



D. Transition Matrix

$$A=\begin{bmatrix} 0 & 0 & R3=a_{13} & R4=a_{14} & R5=a_{15} \\ S1=a_{21} & 0 & 0 & 0 & 0 \\ 0 & S2=a_{32} & 0 & 0 & 0 \\ 0 & 0 & S3=a_{43} & 0 & 0 \\ 0 & 0 & 0 & S4=a_{54} & 0 \end{bmatrix}$$