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Modeling and Assessing the Sustainability of Dams in the United States

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MODELING AND ASSESSING THE SUSTAINABILITY OF DAMS IN THE UNITED STATES

BY

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DEDICATION

To

Pengxian Song

Yanfeng Wang

Maoqian Song

Qiong Yu

And

Yinuo Song

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ABSTRACT

MODELING AND ASSESSING THE SUSTAINABILITY OF DAMS IN THE UNITED STATES

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Dam decision-making is often controversial as a choice has to be made between the benefits provided by dams (e.g., recreation, water supply, hydropower) and their potential negative impacts (e.g., effects on natural flow regime, impediment for fish migration). However, our understandings of such tradeoffs under a full range of dam management alternatives remain limited which hinders our ability to make sound and scientifically defensible dam management decisions. The diverse stakeholders involved in the decision-making process with varying perspectives and preferences could further exacerbate the difficulty of decision-making. To advance our knowledge in sustainable dam decision-making, this dissertation developed modeling tools to evaluate dam decisions based on greenhouse gas (GHG) emissions, hydropower generation, sea-run fish population, and management cost from both spatial and temporal perspectives. The developed model was further applied in role-paly simulation workshops to investigate the potential differences between scientifically optimized decisions and the negotiated consensus. The results revealed that although most hydroelectric dams have comparable GHG emissions to other types of renewable energy (e.g., solar, wind energy), electricity produced from tropical reservoir-based dams could potentially have a higher emission rate than fossil-based electricity. It is possible to simultaneously optimize energy, fish, and cost outcomes through strategic dam management actions. Basin-scale management strategies may

outperform individual dam management strategies because the former can provide a broader set of solutions for balancing complex tradeoffs than the latter. Furthermore, diversification of management options (e.g., combination of fishway installations, dam removals, and generation capacity) may have the highest potential in balancing fish-energy-cost tradeoffs. Finally, dam management negotiation is helpful in facilitating decisions with more balanced outcomes but not necessary reflect the environmentally optimal outcomes.

CHAPTER 1: INTRODUCTION

The US river systems are heavily dammed by more than 2 million dams. These dams provide critical services for human society. For example, hydropower is currently the largest source of renewable energy in the US, accounting for 44% of the total renewable energy generation in 2017 (EIA, 2018a; Song et al., 2018; Uría-Martínez et al., 2015). Reservoirs created by dams provide nearly 40% of the agricultural, domestic, and industrial water demand (Biemans et al., 2011). On the other hand, the adverse impacts induced by dams and their safety risks are increasingly being recognized. For instance, dams have been criticized for altering natural flow regimes, blocking fish passage, affecting sediment transport, and changing watershed characteristics, which collectively contribute to the degradation of water quality, fish population, and biodiversity, resulting in cascading social and economic problems (e.g., revenue loss in the fishing industry) (Bunn and Arthington, 2002; Gehrke, P.C. et al., 2002; Liermann et al., 2012; Poff et al., 2007; Ziv et al., 2012). Furthermore, some of the older and/or larger dams are often perceived as a public-safety risk under the increasing possibility of natural and man-made threats (Hartford and Baecher, 2004; McClelland and Bowles, 2002).

Given the increasing number of dams that have approached or exceeded their design life and shifted their primary functions in the developed countries, many decisions about whether to remove, rehabilitate, or upgrade existing dams need to be made in the coming decades. Dam decision-making, however, is often highly contentious as tradeoffs between giving up their benefits and the avoidance of their negative impacts exist and vary widely depending on the type and context of dam management actions. The difficulty is further exacerbated by the diverse stakeholders involved in the decision-making process whose perspectives and preferences of the outcomes vary significantly. For example, federal agencies such as U.S. National Oceanic and

Atmospheric Administration Fisheries Services (NOAA) and U.S. Fish and Wildlife Service (FWS) may support dam removals for fisheries restoration (Lenhart, 2003), while stakeholders such as hydropower operators or residents who live near the dam usually prefer to keep dams to satisfy their need for hydropower generation or recreational services (Burger, 2011). To facilitate dam decisions that optimize both human benefits and ecosystem sustainability, a holistic understanding of the tradeoffs between multiple interrelated sectors in the dam system is essential.

Nevertheless, our knowledge of complex dam tradeoffs remains limited mainly from the following four aspects. First, previous tradeoff studies have mainly focused on a single type of management practice, including construction (Wild et al., 2018; Ziv et al., 2012), removal (Kuby et al., 2005; Null et al., 2014; Roy et al., 2018), fishway installation (Kuby et al., 2005; Song et al., 2019), or turbine shutdown (Eyler et al., 2016; Song et al., 2019; Trancart et al., 2013). Combinations of multiple dam management strategies were often not considered. Second, there is still a lack of knowledge about the tradeoffs among various environmental, economic, and social outcomes under different dam management actions. Third, temporal and spatial considerations were lacking in previous studies. For example, simplified proxies, such as habitat gains (Null et al. 2014; Roy et al. 2018) and reconnected areas (Kuby et al. 2005; Ziv et al. 2012), have been widely used in previous tradeoff studies to estimate the potential changes in fish populations which do not reflect the temporal changes of fish populations in response to different dam management activities. In addition, contradictory conclusions about whether hydropower is a low-carbon or carbon-intense energy have drawn without considering spatial factors. Fourth, investigation of the potential differences between the negotiated consensus and scientifically optimized decisions was still limited.

To address these knowledge gaps, this dissertation seeks to answer the following six main questions.

Are there optimal dam management strategies that balance tradeoffs between hydropower generation, fish population, and project cost? (Chapter 2 and 3)

How do diversifying dam management options benefit the balance of the energy-fish-cost tradeoffs? (Chapter 2 and 3)

How do individual and basin-scale dam management strategies compare? (Chapter 3)

How do dam life span, upstream passage rate, and downstream habitat area influence the temporal changes of fish population, fish restoration period, and the energy-fish tradeoff? (Chapter 4)

How do lifecycle GHG emissions from dams associated with their spatial locations, types, and scales? (Chapter 5)

How do negotiated decision and scientifically optimal alternatives compare? (Chapter 6)

CHAPTER 2: MANAGING DAMS FOR ENERGY AND FISH TRADEOFFS: WHAT DOES A WIN-WIN SOLUTION TAKE?

2.1. Introduction

Hydropower is currently the largest source of renewable energy in the United States of America (USA), accounting for 44% of the total renewable energy generation in 2017 (EIA, 2018a; Song et al., 2018; Uría-Martínez et al., 2015). This energy is generated by around 2300 hydroelectric dams, with an installed capacity ranging from 50 W to 6495 MW (Samu et al., 2018). An additional 50% increase in generation capacity is expected by 2050 through the conversion of non-powered dams, capacity expansion of existing hydroelectric dams, and construction of pumped storage facilities (DOE, 2016). However, these dams are often cited as a major causal factor in the dramatic decline of fish populations, especially the diadromous fish species that migrate between marine and freshwater habitats to spawn (Brown et al., 2013; Limburg and Waldman, 2009; Trancart et al., 2013; Ziv et al., 2012). For example, alewife landings on the US east coast have declined more than 90% following the construction of a series of dams in the early 20th century (McClenachan et al., 2015; Opperman et al., 2011). Hydroelectric dams affect fish populations both directly and indirectly through turbine injuries (Schaller et al., 2013; Stich et al., 2015), loss of accessible spawning habitat (Hall et al., 2011), and degradation of habitat quality (e.g., changes in temperature, morphology, and discharge) (Johnson et al., 2007). Various management actions such as dam removals (Magilligan et al., 2016; O'Connor et al., 2015), the installation of fish passage structures (hereafter referred to as fishways) (Nyqvist, D. et al., 2017; Schilt, 2007), and periodic turbine shutdowns (Eyler et al., 2016), have been implemented to restore river connectivity and mitigate impacts on diadromous fish species. According to data collected by American Rivers, more than a thousand dams have been removed in the USA in the last two decades (American Rivers, 2017). In cases where hydroelectric dams remain intact, fishways are often installed to assist with upstream and downstream fish migrations (Silva et al., 2018), and have been mandated by the Federal Energy Regulatory Commission (FERC) as part of dam relicensing process since the 1960s (Gephard and McMenemy, 2004). Turbine shutdowns are also employed to reduce mortalities during peak fish downstream migration periods and have been widely applied to lessen injuries and mortality due to blade strikes, pressure changes, and cavitation (Jacobson et al., 2012).

Though these approaches have been useful in lessening the impacts of hydropower operation on diadromous fish species, a loss of hydropower generation is inevitable in all three practices (Gatke et al., 2013; Null et al., 2014; Trancart et al., 2013). For example, a loss of $57 million annual hydropower revenue resulted from the removal of the Shasta Dam in California's Central Valley, though this removal reopened around 1700 km of upstream salmonid habitat (Null et al., 2014). Fishway installations reduce hydropower production by diverting water discharge to fish passage structures (Gatke et al., 2013). Power cannot be generated during turbine shutdowns. From the perspective of the dam operator, carefully planning of shutdown periods to maximize downstream migrant survival is important to minimize hydropower generation losses (Trancart et al., 2013).

Though researchers and decision-makers have widely recognized energy-fish tradeoffs, quantification of such tradeoffs to inform the decision-making process remains limited (Lange et al., 2018). Simplified proxies, such as habitat gains (Null et al., 2014) and reconnected areas (Kuby et al., 2005), are widely used to estimate the potential increase of fish populations. However, these methods largely neglect factors such as the effectiveness of dam management strategies on both upstream and downstream passage, environmental capacities of reopened habitats, and other dynamics within the entire fish life cycle (Godinho and Kynard, 2009; Sweka et al., 2014; Ziv et al., 2012). Structured fish population models are another means to quantitatively simulate fish populations by considering and incorporating different mortality sources at each of the individual fish life cycle stages. Previous studies have developed and applied structured population models to assess the effect of dam passage rates on diadromous fish populations (Burnhill, 2009; Nieland et al., 2015; Stich et al., 2018). However, this method has not been used to explore the energy-fish tradeoffs of dam management. Furthermore, these studies run on annual or monthly time steps and could not capture the effect of turbine shutdowns that only operate for several days or weeks during peak migration (Trancart et al., 2013).

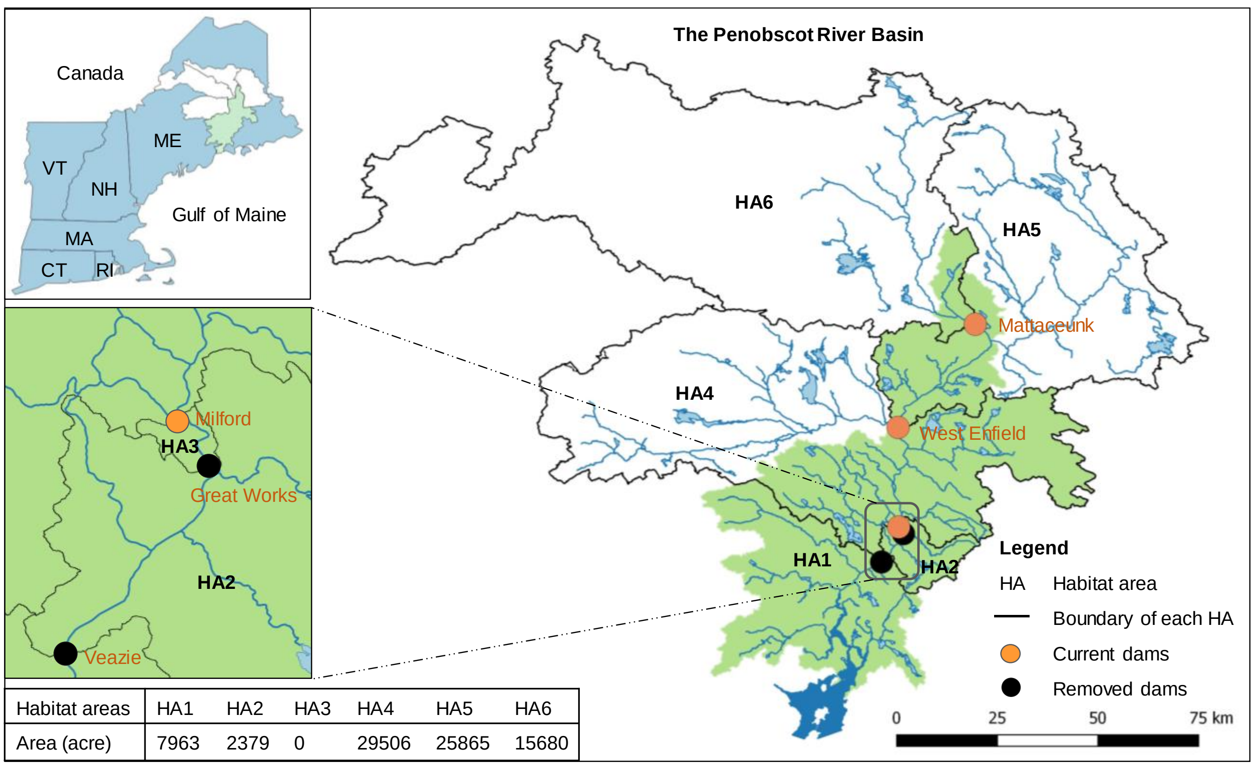
In river systems with multiple dams, regional or basin-scale approaches are preferred over site-specific approaches because of the cumulative effect of dam passage on migrants moving further upstream (Neeson et al., 2015; Opperman et al., 2011; Winemiller et al., 2016). Basin-scale outcomes under various dam management practices could differ dramatically as hydropower potential and fish habitats are unevenly distributed (Roy et al., 2018). However, many previous studies exploring energy-fish tradeoffs on a regional scale have focused on only a single type of management practice (e.g., dam removal or construction) rather than comparing multiple different strategies. For instance, a new dam construction project in the Mekong River Basin was investigated by Ziv et al (2012) to understand the tradeoffs between hydropower production, migratory fish biomass, and fish diversity using the production possibility frontier method (Ziv et al., 2012). Null et al. (2014) analyzed tradeoffs between habitat gains and hydropower generation under dam removal scenarios in California's Central Valley using an economic-technical optimization model (Null et al., 2014). Trancart et al (2013) optimized the timing and duration of turbine shutdowns that would avoid 90% loss of European Eels during seaward migration on the Oir River, France, by forecasting eels' migration peaks based on an auto-regressive integrated moving average model (Trancart et al., 2013). Only one study, conducted in the Willamette basin, Oregon, simulated both dam removal and fishway installation to co-optimize their effects on salmon and hydropower generation (Kuby et al., 2005). This study concluded that fishway installations could be as effective as dam removals at connecting upstream and downstream habitat. However, this study did not measure the actual effectiveness of the fishways, which were treated as either entirely passable or not passable for salmon. The effect of turbine fish kills during downstream migration was also neglected.

The limited consideration of multiple dam management options and important fish mortality factors could potentially lead to sub-optimized decision-making (Sweka et al., 2014). Accordingly, this study developed a system dynamics modeling (SDM) framework to investigate the tradeoffs between hydropower generation and potential diadromous fish abundance. SDM is a computational method using a set of linked differential equations to simulate the behavior of complex systems over a certain time period. It is grounded in system thinking and has been widely recognized as a powerful tool to study interactions among system components through capturing system feedback loops and delays (Forrester, 1997; Sterman, 2001). SDM has been previously applied to simulate hydropower production (Bosona and Gebresenbet, 2010; Sharifi et al., 2013) and fish abundance (Barber et al., 2018; Ford, 2000; Stich et al., 2018), but it has not been used to explore the tradeoffs between these two sectors. In this study, the developed framework was used to investigate the potential of three different dam management practices, including dam removals, fishway installations, and periodic turbine shutdowns. Four critical questions regarding dam management were asked, including (1) how and to what extent does each dam management practice influence the energy-fish tradeoffs? (2) what might be the best dam management solution in minimizing energy loss and maximizing fish population on a basin scale? (3) how do upstream and downstream passage rates influence population abundance? and (4) what are the key determinants in managing the dam related energy-fish tradeoffs?

2.2. Material and Methods

2.2.1. Model river description

The model framework assessed for decision-making was based on an abstraction of the Penobscot River, Maine, which is the second largest river system in the northeast USA, with a drainage area of approximately 22,000 km^2 (Izzo et al., 2016; Trinko Lake et al., 2012). This large river system historically provided important spawning and rearing habitat for 11 native diadromous fish species that have high commercial, recreational, and ecological value to local communities (Kiraly et al., 2015). Among these species, alewives (*Alosa pseudoharengus*) have been a major source of traditional river fisheries since the beginning of human settlement in the region (McClenachan et al., 2015). Alewives are small anadromous fish that have high rates of iteroparity (reproduce multiple times over their lifetime) in Maine. Alewives are also the base of marine, freshwater, and terrestrial food webs (ASMFC, 2009). Changes in their abundance may also influence the population dynamics of their predators, including the endangered Atlantic salmon (*Salmo salar*) (Lichter et al., 2006). From 1634 to 1900, industrial dams were heavily developed on the Penobscot River, and little or no access to spawning habitat was later identified as the main cause for the alewife population crash during that period (McClenachan et al., 2015). Alewife habitat areas (HAs) are unevenly distributed among the river segments created by the dams (Figure 2-1). A much larger amount of HA is located upstream of the Milford Dam than downstream of it. Restoration efforts began in the 1940s to combat diadromous fish declines (Rounsefell and Stringer, 1945). One of the largest efforts was the Penobscot River Restoration Project (PRRP), which from 2012 to 2013 removed the two dams furthest downstream and improved fish passages at the remaining dams (Figure 2-1) (Opperman et al., 2011). To test the effectiveness of the PRRP and alternative basin-scale dam management strategies, the five run-of-river hydroelectric dams historically on the main-stem of the river was chosen to study, which from downstream to upstream included Veazie, Great Works, Milford, West Enfield, and Mattaceunk dams (Table 2-1 and Figure 2-1). Dams located on the tributaries were ignored for simplification.



Habitat areas	HA1	HA2	HA3	HA4	HA5	HA6
Area (acre)	7963	2379	0	29506	25865	15680

Figure 2-1. Map of the study area showing the locations of the five hydroelectric dams as well as current and historic alewife spawning lakes/ponds in the Penobscot River Basin. The inserts show the Penobscot River basin within the northeastern USA (upper map) and the partial Penobscot River main-stem from Veazie to Milford Dam (lower map).

Table 2-1. Project information for the five studied dams in the main-stem of the Penobscot River, Maine.

Dams[a] (distance to ocean)	Year	Installed capacity[b] (Amaral et al., 2012) (MW)	Turbine's maximum flow (Amaral et al., 2012) ($\times 10^6$ m^3/d)	Rated head (Amaral et al., 2012)	Dam length (USACE, 2016) (m)	Dam height (USACE, 2016) (m)	Upstream passage facilities (Amaral et al., 2012)	Potential downstream passage routes (Amaral et al., 2012)
Veazie[c] (Dam 1) (rkm 55)	1912	9.3	13.6	7.3	257	10	One vertical slot fishway	Sluice gate, turbine units and spillway
Great Works[c] (Dam 2) (rkm 69)	1900	7.6	21.1	5.3	331	6.1	Two Denil fishways	Bypass pipe (2000), 3 gated outlet[d], turbine units, and spillway
Milford (Dam 3) (rkm 73)	1906	8.0	17.2	5.8	426	10	One Denil fishway, one fish elevator (installed in 2014)	Log sluice gate[e], turbine units, and spillway
West Enfield (Dam 4) (rkm 114)	1894	25.4	22.0	7.9	296	14	One vertical slot fishway, one Denil fishway (backup fishway)	Gated section, turbine units and spillway
Mattaceunk (Dam 5) (rkm 175)	1939	21.6	18.2	11.9	357	14	One pool and weir fishway, one fishlift	Bypass system, roller gate, debris sluice gate, turbine units and spillway

Notes:
[a] – All five dams are run-of-river dams. The primary function of these dams is hydropower generation;
[b] – Installed capacity refers the maximum output of electricity that a generator can produce under ideal conditions (EIA, 2019);
[c] – Veazie Dam and Great Works Dam have been removed in summer 2013 and 2012, respectively.
[d] – The 3 gated outlets are currently used to increase discharge capacity under flood conditions rather than downstream fish passage;
[e] – The 3-meter wide gate is used as downstream bypass at the Milford dam. The gate flow is set at 3 m^3/s during the established migration periods.

2.2.2. Integrated energy and fish population model

An integrated energy-fish model that couples hydropower generation and age-structured fish population models was used to analyze the tradeoffs between energy and fish abundance under various dam management scenarios at a basin scale. The energy-fish model was built in Vensim® DSS, one of the most widely used platform for SDM (Ford, 2000). Figure 2-2 presents an abstracted version of the stock-and-flow diagram of SDM model developed in this study. The energy model and the age-structured fish population model are integrated through three dam management practices, including fishway installations, turbine shutdowns, and dam removals. A complete version of the model is provided as a Vensim file in the Appendix A. The model runs across 150 years on a daily time step to ensure stabilization.

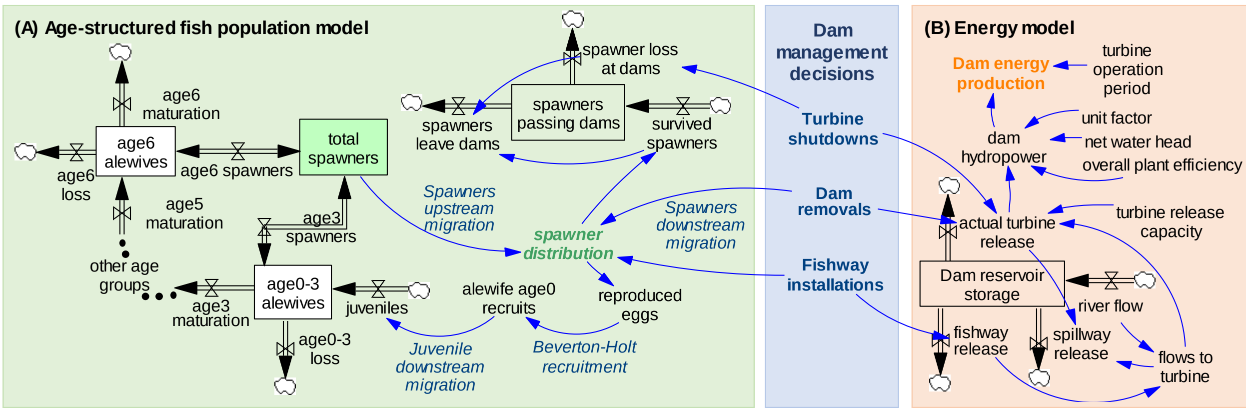


Figure 2-2. An abstracted stock-and-flow diagram showing the key components of the age-structured fish population model (A) and the hydropower generation model (B).

Hydropower generation

Hydroelectric dams convert the natural flow of water into electricity when falling water turns the blades of a turbine connected to a generator. The general equation for hydropower generation (Adeva Bustos et al., 2017; Hadjerioua et al., 2012; Power, 2015; Singh and Singal, 2017) is:

$$E = P \times t = Q \times H \times \eta \times \rho \times g \times 10^{-6} \times t \qquad \text{Equation 2-1}$$

where E is the generated energy, MWh; P is the power produced at the transformer, MW; t is turbine operation period, hours; Q is the volume flow rate passing through the turbine, m^3/s; H is the design net head, meters; η is the overall efficiency, assumed to be 0.85 (Hadjerioua et al., 2012; Power, 2015); ρ is the density of water, 1,000 kg/m^3; and, g is the acceleration due to gravity, 9.8 m/s^2.

Given that run-of-river dams do not have large reservoirs and generally have limited impacts on river flows, the total water inflow was assumed to always be equal to the total outflow for each dam. Evaporation and system leakages were assumed to be zero. At hydropower dams, river flow is diverted to different paths following a minimum flow discharge rule (Basso and Botter, 2012; Lazzaro et al., 2013). First, a portion of the water is diverted to meet the operation needs of the fish passage structures, including ensuring that fish will be attracted to the fishways. Two approaches have been reported for determining fishway attraction flow, including 1-5% of the mean annual streamflow (Bolonina et al., 2016) and at least 5% of powerhouse hydraulic capacity (USFWS, 2017). In this study, we use the larger value between 5% of mean annual river flow and 5% of the maximum turbine release capacity as fishway attraction flow. The remaining water was then assumed to be available for hydropower generation. The actual amount of water releasing from turbine facilities is determined by the remaining water flow in the river, the turbine's minimum admissible flow rate, and its maximum flow rate. If the remaining water flow is less than the turbine's minimum admissible flow rate, all of the remaining water flow will be released from the spillway. If the remaining water flow is greater than the turbine's maximum flow rate, water volume in excess of the maximum flow rate will also be released from the spillway. Otherwise, all remaining water will be released from the turbines.

We used the drainage-area ratio method to extrapolate the river inflow of all five hydroelectric dams from the daily streamflow data obtained from two U.S. Geological Survey gages (01034500 Penobscot River at West Enfield and 01034000 Piscataquis River at Medford (USGS, 2001-2015)) for the period of January 2001 to December 2015. The detailed calculation process at each dam site is provided in Section A1 of the Appendix A. This calculated daily river inflow in a 15-year time period was then repeated and expanded to 150 years. The maximum turbine flow rate at each studied dam was collected from the related reports (Table 2-1) (Amaral et al., 2012; Great Lakes Hydro America LLC, 2016). The minimum admissible flow rate was assumed to be 40% of the maximum flow (Power, 2015). The design net head at each dam was assumed to be equal to the rated head of installed turbines obtained from Amaral et al (2012) (Table 2-1). Turbine units only operate when river discharge satisfies turbines' hydraulic capacities (Power, 2015). The influence of market demand on hydropower generation was ignored.

Age-structured fish population model

The daily age-structured alewife population model used in this study was adapted from a yearly age-structured model presented in Barber et al (2018). Alewife abundance was simulated by keeping track of the activities and survivals of different age groups on a daily stepwise progression (Figure 2-3). Alewives mature between the ages of three and eight. The probabilities in reaching sexual maturity at different ages were obtained from (Gibson and Myers, 2003) and (Barber et al., 2018). The matured alewives migrate upstream to freshwater habitats to spawn between March and June (Eakin, 2017; Hasselman et al., 2014; Rosset et al., 2017). After spawning, surviving adults return to the ocean. Low dam passage rates for fish migrating upstream can affect accessibility to spawning habitat. Dams can also cause migratory delays and increased mortality rates for spawners moving both upstream and downstream, which can potentially result in a population decline. Dam passage rates were explicitly modelled in this study. In freshwater spawning habitat, eggs hatch into larvae and grow to juveniles. Juveniles move downstream between mid-July and early December, and can also experience dam-related delay and mortality during their migration. The surviving juveniles enter the ocean and continue to grow until reaching sexual maturity, thus completing the cycle. Alewives generally survive up to 9 years in the wild. In our model, alewives older than 6 years were not included in simulations because these age groups only account for around 5% of the total spawner population (Messieh, 1977). Alewife activities such as spawner upstream migration, egg production, and post-spawner and juvenile downstream migration were assumed to happen once every year on designated days. The detailed equations are provided below.

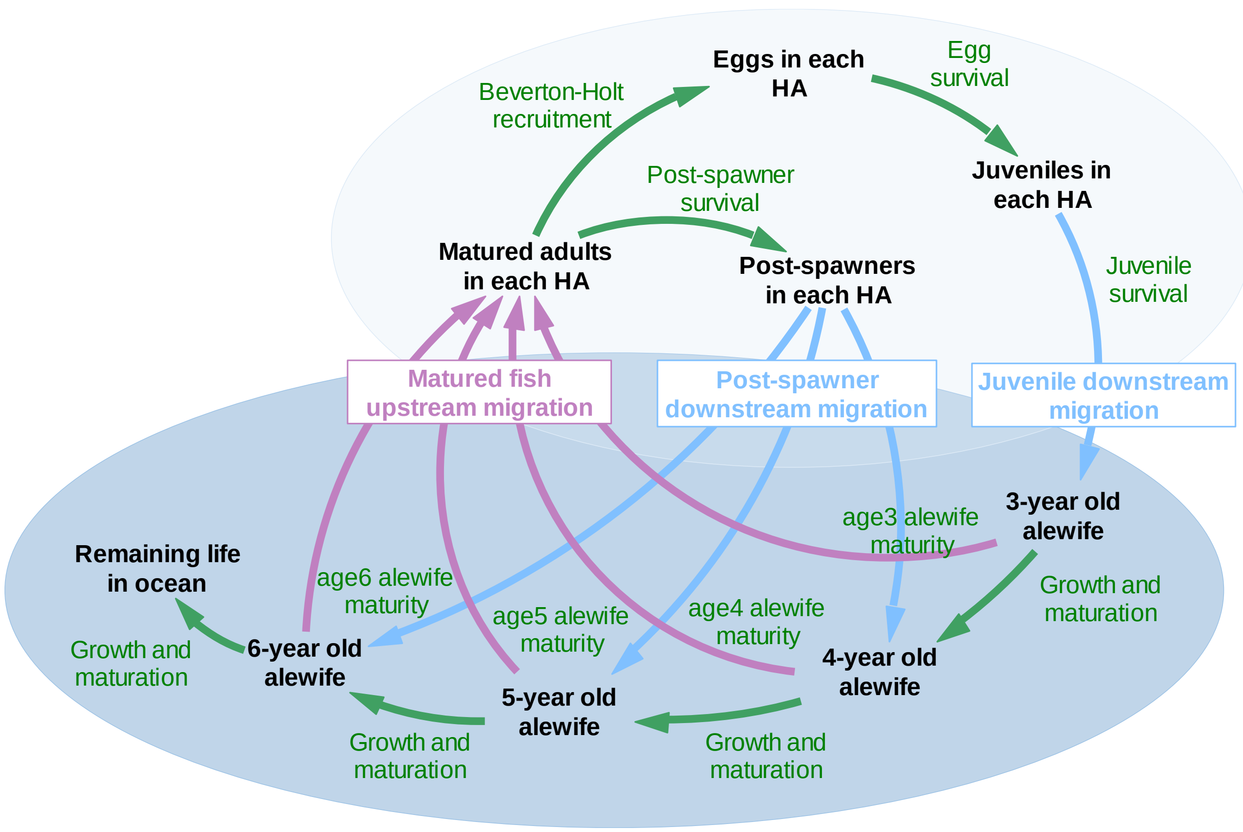


Figure 2-3. Life stages of alewife included in the age-structured fish population model. The light and dark blue ellipses refer to the freshwater and ocean habitats of alewife, respectively.

For a given spawning period, the number of eggs produced in each HA is a function of females that survived to spawn in that area and their fecundity:

$$E_{HA_j,t,a} = \sum_{i=3}^{6}(S_{HA_j,i,t,a} \times r_{F:M} \times \varphi \times F_i) \qquad \text{Equation 2-2}$$

where, $E_{HAj,t,a}$ is egg production of alewife in HA_j (j =1-6) for a given year t on the a^{th} day (a was assumed to be May 20th, the 140th day of each year (Rosset et al., 2017)), millions; $S_{HAj,i,t,a}$ is the total number of surviving age-i alewife to spawn at HA_j in year t on the a^{th} day, millions; $r_{F:M}$ is female to male ratio that was assumed to be 0.5 (Barber et al., 2018); φ is the probability of spawning, 0.95 (Barber et al., 2018); and, F_i is the fecundity of age-i alewife which was assumed to be linearly related to the mass of age-i alewife (Table S1).

Juvenile production was modeled as a density-dependent process, which was characterized using the Beverton-Holt spawner-recruit (B-H) curve (Equation 2-3). The B-H curve was chosen for this model because a study of eight alewife populations in the northeast region of the USA indicated it was a better fit than the Ricker curve (Barber et al., 2018; Gibson, 2004).

$$J_{HA_j,t,b} = \frac{\alpha \times E_{HA_j,t,a}}{1+\frac{\alpha \times E_{HA_j,t,a}}{A_j \times R_{asy}}} \quad \text{Equation 2-3}$$

where $J_{HAj,t,b}$ is the number of juveniles at HA_j at the beginning of the downstream migration for a given year t on the b^{th} day (b was assumed to be August 18th, the 230th day of each year (Iafrate and Oliveira, 2008; Yako et al., 2002)), millions; R_{asy} is the asymptotic recruitment level, which indicates the carrying capacity of freshwater habitats expressed as the amount of survived juveniles per acre, 3283 age-0 fish/acre (Barber et al., 2018); α is the lifetime reproduction rate of alewife, 0.0015 (Gibson, 2004); A_j is the size of HA_j (j =1-6), acres.

During downstream migration, juveniles pass each dam through one of three routes: the spillway (or sluiceway), the fish bypass system, or a turbine (Schilt, 2007). The partitioning of alewives to each route was based on the relative amount of water being released through each route at a given time step (Nyqvist, Daniel et al., 2017). Other factors that could potentially affect fish distributions, including installation of screening system and sensory stimuli (e.g., light (Johnson et al., 2005; Mueller et al., 2001), sound (Nestler et al., 1992), turbulence (Coutant, 2001), and electric fields (Schilt, 2007)) were not considered. Turbine mortality rates were assumed to be 30% when in operation and 0% during shutdowns (Pracheil et al., 2016). The other two migration routes are generally considered benign (Muir et al., 2001; Stich et al., 2014) and the simplifying assumption was made that their mortality rates were zero. The number of juveniles entering the ocean was determined by the cumulative turbine mortality (Equation 2-4).

$$J_{ocean,t,c} = \sum_{j=1}^{6}(J_{HA_j,t,b} \times \prod_{k=1}^{j-1} \frac{Q_{turbine_k,t,c}}{Q_{dam_k,t,c}} \times (1 - M_{turbine_k})) \quad \text{Equation 2-4}$$

where $J_{ocean,t,c}$ is the number of surviving juveniles entering ocean in year t on the last day of the downstream migration period c (c was assumed to be the 240th day of each year), millions; $Q_{turbine_k,t,c}$ and $Q_{dam_k,t,c}$ are the turbine and the total water flow rate of Dam k (k =1-5) in year t on the c^{th} day, respectively, m^3/d; $M_{turbine_k}$ is the turbine mortality rate of Dam k, 0.3 (Pracheil et al., 2016) during operation and 0 during turbine shutdowns.

In the ocean, immature alewives between ages 2 and 6 have a probability of reaching sexual maturity and entering the spawning run the next year. Alewife maturity at each age is provided in Table S1. The population of age-i fish in the ocean in year t, $O_{i,t,d}$, was calculated based on the populations of both immature fish, $NS_{i,t,d}$, and mature fish, $S_{i,t,d}$ (Equation 2-5) where d denotes the beginning of each fish upstream migration period, which was assumed to be the 120th day of each year (Chadwick and Claytor, 1989; Ellis and Vokoun, 2009).

$$O_{i,t,d} = NS_{i,t,d} + S_{i,t,d} \quad \text{Equation 2-5}$$

Immature fish remain in the ocean, and their abundance was calculated by applying an annual ocean mortality rate (including all natural causes of death in the ocean), M_{ocean} (assumed to be 0.648 (Barber et al., 2018)), on the d^{th} day every year, and the probability of maturation at each age, m_i (Equation 2-6 and Table S1). The abundance of age-0 immature fish, $NS_{0,t,d}$, was assumed to be equal to juveniles entering the ocean, $J_{ocean,t,c}$.

$$NS_{i,t,d} = NS_{i-1,t-1,d} \times e^{-M_{ocean}} \times (1 - m_i) \quad \text{Equation 2-6}$$

The mature fish stock in the ocean (Equation 2-7) included first-time spawners, $S_{i,t,0,d}$ (calculated in Equation 2-8) and repeat spawners, $S_{i,t,p,d}$.

$$S_{i,t,d} = S_{i,t,0,d} + \sum_p S_{i,t,p,d} \quad \text{Equation 2-7}$$

$S_{i,t,0,d} = NS_{i-1,t-1,d} \times e^{-M_{ocean}} \times m_i$ Equation 2-8

Repeat spawners have spawned at least one time and are subject to natural (i.e., predation, delayed migration, or senescence), fishing (both commercial and recreational), and other anthropogenic (i.e., turbine) mortalities. Natural mortality included both ocean mortality and spawning mortality, with the latter incorporating all natural causes of death in freshwater. For a given spawning run, the total number of spawners reaching the suitable habitat areas was calculated using Equation 2-9.

$\sum_{j=1}^{6} S_{HA_j,t,a} = S_{t,d} \times (1 - M_{fishing}) \times (1 - M_{spawn})$ Equation 2-9

where, $S_{HAj,t,a}$ is the number of spawners at HA_j that are ready to spawn in year *t*, millions; $S_{t,d}$ is the abundance of mature fish in the ocean before the spawning run in year *t*, millions; $M_{fishing}$ is the interval fishing mortality, 0.4 (Barber et al., 2018; MaineDMR, 2016); M_{spawn} is the interval spawning mortality associated with each spawning run, 0.45 (Barber et al., 2018; Durbin et al., 1979; Kissil, 1974). The spawning run was assumed to last 30 days with upstream migration, spawning, and downstream migration each taking 10 days (Frank et al., 2011; Franklin et al., 2012).

The value of $S_{HAj,t,a}$ was determined by the cumulative upstream passage rate of dams downstream of HA_j as well as a dispersal rule. In this study, upstream passage rate was defined as the percentage of individuals that are attracted to, enter, and successfully ascend a fishway (Silva et al., 2018). Alewives have a tendency to return to their natal area to spawn (McBride et al., 2014; Pess et al., 2014). Accordingly, two dispersal rules were investigated in this study to investigate two opposing conditions related to fish dispersal. The first rule assumed that alewife distribution was based on the habitat size of the entire basin despite the influence of dam structures. The second rule took into account the long-term blockage effect of dams which restricts alewives' motivation to seek habitats that were suitable for spawning but no longer accessible. Equation 2-10 and 2-11 describe the calculations of the two dispersal rules.

If $\frac{A_j}{A} > D_{HA_j}, S_{HA_j,t,a} = \left(\frac{A_j}{A} + \left(D_{HA_j} - \frac{A_j}{A}\right) \times \left(1 - P_j\right)\right) \times \sum_{j=1}^{j=6} S_{HA_j,t,a}$ Equation 2-10

If $\frac{A_j}{A} \leq D_{HA_j}, S_{HA_j,t,a} = D_{HA_j} \times \sum_{j=1}^{j=6} S_{HA_j,t,a}$ Equation 2-11

where, A_j is the size of HA_j (j = 1-6), acres. The size of each HA was estimated as the summed acreage of the documented alewife spawning ponds within each river segment, obtained from the Maine Stream Habitat Viewer provided by the Maine Department of Marine Resources Coastal Program (MaineDMR, 2017). A was the total habitat area, which equaled 81,393 acres when alewives were homing to the entire basin under the first dispersal rule or the sum of HAs used by alewives (based on results obtained from the first dispersal rule) under the second dispersal rule. D_{HA_j} was a dispersal factor that was calculated using Equation 2-12.

$D_{HA_j} = (D_{HA_{j-1}} - \frac{A_{j-1}}{A}) \times P_{j-1}$ Equation 2-12

D_{HA_1} = 1. P_j is the upstream passage rate of the j^{th} dam. P_j was assumed to be 0 when no fishway was present and 0.7 (Bunt et al., 2012; Noonan et al., 2012) when fishways were present.

Shortly after spawning, post-spawners migrate seaward and encounter turbine and ocean mortalities prior to their next spawning run. The abundance of repeat spawners in the ocean at the beginning of upstream migration was calculated using Equation 2-13 (Table S1).

$$S_{i+1,t+1,p+1,d} = \sum_{j=1}^{6}(S_{HA_j,i,t,p,a} \times \prod_{k=1}^{j-1} \frac{Q_{turbine_k,t,c}}{Q_{dam_k,t,c}} \times (1 - M_{turbine_k})) \times e^{-0.92 M_{ocean}}$$

Equation 2-13

where, the annual ocean mortality, M_{ocean}, was prorated to 0.92 indicating that 335 out of 365 days, spawners live in the ocean and are subject to ocean mortality.

A few additional assumptions were made for simplification. Alewives at each age were assumed to experience the same delay time as well as ocean and spawning mortality rates during both downstream and upstream migrations. The carrying capacities of each unit of habitat area were assumed to be the same. The influence of temperature on the timing of upstream migration and spawning was ignored.

2.2.3. Model validation and sensitivity analysis

Behavior test

Once values for the parameters of the integrated model were selected, the accuracy of the model was tested through a behavior test. For the energy model, annual hydropower generation at Milford and West Enfield dams were calculated and compared with the historical data (2001-2015) obtained from the U.S. Energy Information Administration (EIA, 2018b). The correlation coefficient (r^2) was used to test the goodness of fit between simulated and historical yearly hydropower generation. Correlation was relatively high, with a calibrated r^2 of 0.60 for Milford Dam and 0.86 for West Enfield Dam (Section A3 of the Appendix A).

The behavior test of the fish model was conducted by checking that the simulated fish abundance entering the Penobscot River was within the range of total alewife abundance entering rivers in Maine. Total abundance for the state of Maine was calculated based on Alewife landings data (in million pounds, 1950-2016) collected from the Department of Marine Resources (DMR) (MaineDMR, 2018), average alewife spawner weights (in pound, 0.4 (Barber et al., 2018)), and alewife harvest rates which were assumed in the range of 10-70% (Barber et al., 2018; MaineDMR, 2016). Additionally, the DMR also provided alewife trap counts at the Milford Dam, which were compared against the simulated results at the Milford Dam. Our fish model was initialized with 1 million juveniles entering the ocean. The results showed that the simulated number of alewife spawners after model stabilization was within the range of the historical data (Section A4 of the Appendix A). Additionally, the abundance of simulated spawners passing through Milford dam compared with the trap counts at the same location was within 5-84% difference.

Sensitivity analysis

Sensitivity analysis was conducted to determine which input parameters have the biggest influence on system behavior (Sterman, 1984). We assessed the sensitivity of alewife spawner abundance and hydropower generation to all constant variables within the model. The tested variables related to alewife spawner abundance include spawning mortality, fecundity (slope and intercept), B-H curve related variables (alpha and asymptotic recruitment level), probability of maturity, sex ratio, total habitat area in the basin, turbine mortality, fishing mortality, ocean mortality, and fishway passage rate. The tested variables that are related to hydropower generation include net head, overall efficiency, and turbine operation period. Selected inputs were tested for changes between ±10% and ±90% to capture their practical low and high values. However, a narrower range (e.g., −90 to 50% changes in ocean mortality) was applied when the extreme values became unrealistic. A sensitivity index was calculated for each input change using Equation 14 (Barber et al., 2018; Zhuang, 2014).

$$S = \frac{\frac{O_i - O_b}{O_b}}{\frac{I_i - I_b}{I_b}} \quad \text{Equation 2-14}$$

where O_i is the output value after the input was changed; O_b is the base output value; I_i is the altered input value; and I_b is the original input value. Inputs were considered “highly sensitive” if $|S| > 1.00$.

2.2.4. Dam management scenarios

Eight scenarios were designed to compare the effectiveness of different dam management practices (Table 2-2). The PR-PF-S scenario approximates the PRRP's dam management strategy. Turbine shutdown periods were assumed to be 20 days each year which occur during the 141^{th}-150^{th} day and the 231^{th}-240^{th} day corresponding to the assumed peak downstream migration periods of adults and juveniles, respectively.

Table 2-2. Descriptions of the eight basin-scale dam management scenarios.

Scenarios	Descriptions
NR	All five dams remained in place and no fishway or turbine shutdown was used
PF	Fishway installations at the two most downstream dams
PF-S	Fishway installations and turbine shutdowns at the two most downstream dams
F	Fishway installations at all five dams
F-S	Fishway installations and turbine shutdowns at all five dams
PR-PF	Removal of the two most downstream dams, and fishway installations at the remaining three dams
PR-PF-S	Removal of the two most downstream dams, as well as fishway installations and turbine shutdowns at the remaining three dams
R	All five dams were removed

The influence of upstream and downstream passage efficiency on spawner abundance was further investigated under the F scenario. We assumed upstream passage efficiency to be uniform for all five studied dams and explored changes from 0, 20, 40, 60, 80, and 100% successful passage for each simulation. The same assumption was made for both juvenile and adult downstream passage efficiency.

2.3. Results and Discussions

2.3.1. Energy-fish tradeoffs under various dam management scenarios

We chose alewife spawner abundance as an indicator to show the potential changes of the total alewife populations, as spawners are the main source of fishery (Havey, 1961). The tradeoffs between annual hydropower generation and the stabilized alewife spawner abundance each year under the eight basin-scale dam management scenarios are presented in Figure 2-4. A comparison between the NR and R scenarios show that the five dams can reduce the alewife abundance by 90%. On the other hand, an average of 427 GWh of annual hydropower generation will be lost when all dams are removed, which is around 14% of the annual hydropower generation in Maine (EIA, 2018b).

The performance of fishway installations is heavily influenced by the amount of accessible upstream habitat, the dam mortalities, and the dispersal rules. For instance, in the PF scenario a 30% increase in the total habitat area can lead to a 35% decrease in spawner abundance when spawners home to the entire basin (the first dispersal rule), or a 16% increase when spawners only home to accessible habitats (the second dispersal rule). The decrease of spawner abundance under the first dispersal rule is related to the extremely small sizes of HA2 and HA3. Under this dispersal rule, most spawners have the motivation to move upstream. As Dam 3 is entirely impassible under the PF scenario, this homing instinct result in large amounts of spawners (63%) cumulating in HA2 and HA3 and competing for limited resources, which eventually leads to a reduced survival rate (Section A7 of the Appendix A). Furthermore, as turbines are still in operation in the PF scenario, significant turbine kills could occur when post-spawners and juveniles migrate downstream. In this case, fishways could work as ecological traps and potentially cause a further collapse of the regional fishery (Pelicice and Agostinho, 2008).

Taking the F scenario as another example, the entire watershed becomes accessible to spawners in this scenario, and spawners will mainly be distributed across the four most downstream HAs because HA4 is large enough to support the limited amount of spawners that could successfully pass Dams 1-3. Although the combined size of HAs 1-4 in the F scenario is four times larger than the NR scenario, only a roughly 45% increase in the stabilized spawner abundance is observed. This is due to the high downstream mortality resulting from turbine kills. When turbine shutdown is in operation, an additional 114-134% increase in spawner abundance could be observed (compared to the F-S scenario). When the two most downstream dams are removed (Scenario PR-PF-S), the downstream mortality is further reduced. Hence, an increase of 300-338% of spawner abundance is observed when comparing the PR-PF-S and F scenarios.

The effect of the two dispersal rules is the most prominent in the PF and the PF-S scenarios with a 40-56% difference in spawner abundance. The alewife spawner abundance is lower under the first dispersal rule, as compared to the second one. This is a combined effect of spawner behavior under the two dispersal rules and the availability of the HAs. Unlike the first dispersal rule where spawners moving upstream are mainly driven by homing instincts, under the second dispersal rule spawners moving upstream are mainly driven by competition for resources, and hence the general motivation of moving upstream is comparatively weaker. In this case, the resources in HAs 1-2 could be maximally utilized, resulting in higher spawner abundance. Conversely, under the F, F-S, PR-PF, and PR-PF-S scenarios, alewife spawner abundance is slightly higher under the first dispersal rule than the second one. This is because under these scenarios, a much larger habitat area becomes open and a stronger motivation of moving upstream facilitates spawners reaching the reopened critical habitat. Note, however, that the impacts of dispersal rules on spawner population are marginal (within 2-10% difference) in these scenarios.

If turbine shutdowns reduce mortality as assumed, this approach would be an effective way of lessening fish kills during downstream migration. A comparison of the three scenario pairs (PF vs. PF-S, F vs. F-S, PR-PF vs. PR-PF-S) shows that turbine shutdowns during fish peak downstream migration periods could increase spawner abundance by around 8-30%, 114-134%, and 78-92%, respectively, with small losses of hydropower capacity (~5%). Based on our results, turbine shutdown is the most effective when applied to the F scenario, where the cumulative turbine mortalities associated with three dams (Dams 1-3) are significantly reduced. When turbine shutdowns are applied to the PF or PR-PF scenarios, turbine mortalities associated with two dams (Dams 1 and 2 in the PF scenario and Dams 3 and 4 in the PR-PF scenario) are significantly reduced. As the PR-PF scenario has a much larger size of accessible upstream habitat than the PF scenario, a larger spawner population could benefit from turbine shutdowns and lead to a higher effectiveness of fish restoration. In general, the effectiveness of turbine shutdowns is highly dependent upon spawner dispersal among the habitats, size and location of the accessible HAs, and the number of dam structures that alewives need to traverse in the freshwater environments. Besides, the timing, frequency, magnitude, and duration of seaward migrants each year are also important to the effectiveness of turbine shutdowns (Trancart et al., 2013).

In terms of the energy-fish tradeoffs, the R scenario is the most effective in restoring fish abundance, but would result in the total loss of hydropower capacity. The PF, PF-S, and F scenarios resulted in negligible energy losses, but effects on the spawner abundance are marginal or even negative. The F-S and PR-PF scenarios are able to preserve around 60-92% of the overall hydropower capacity, but only restore spawner abundance to around 35% of the undammed condition. The PR-PF-S scenario, on the other hand, is effective in restoring the spawner population to around 60% of the abundance in the R scenario, with only around a 37% loss of energy. The PR-PF-S scenario also closely reflects the actual management decisions enacted through the PRRP. This project also upgraded hydropower capacity at two tributary dams, which further compensated for energy losses through the removal of the two lowermost dams. Our results suggest that energy-fish tradeoffs could be balanced through utilizing multiple dam management activities at a basin scale. Although dam removal alone is the best option for fish restoration, the resulting hydropower losses could be undesirable in places where hydropower is an important source of energy.

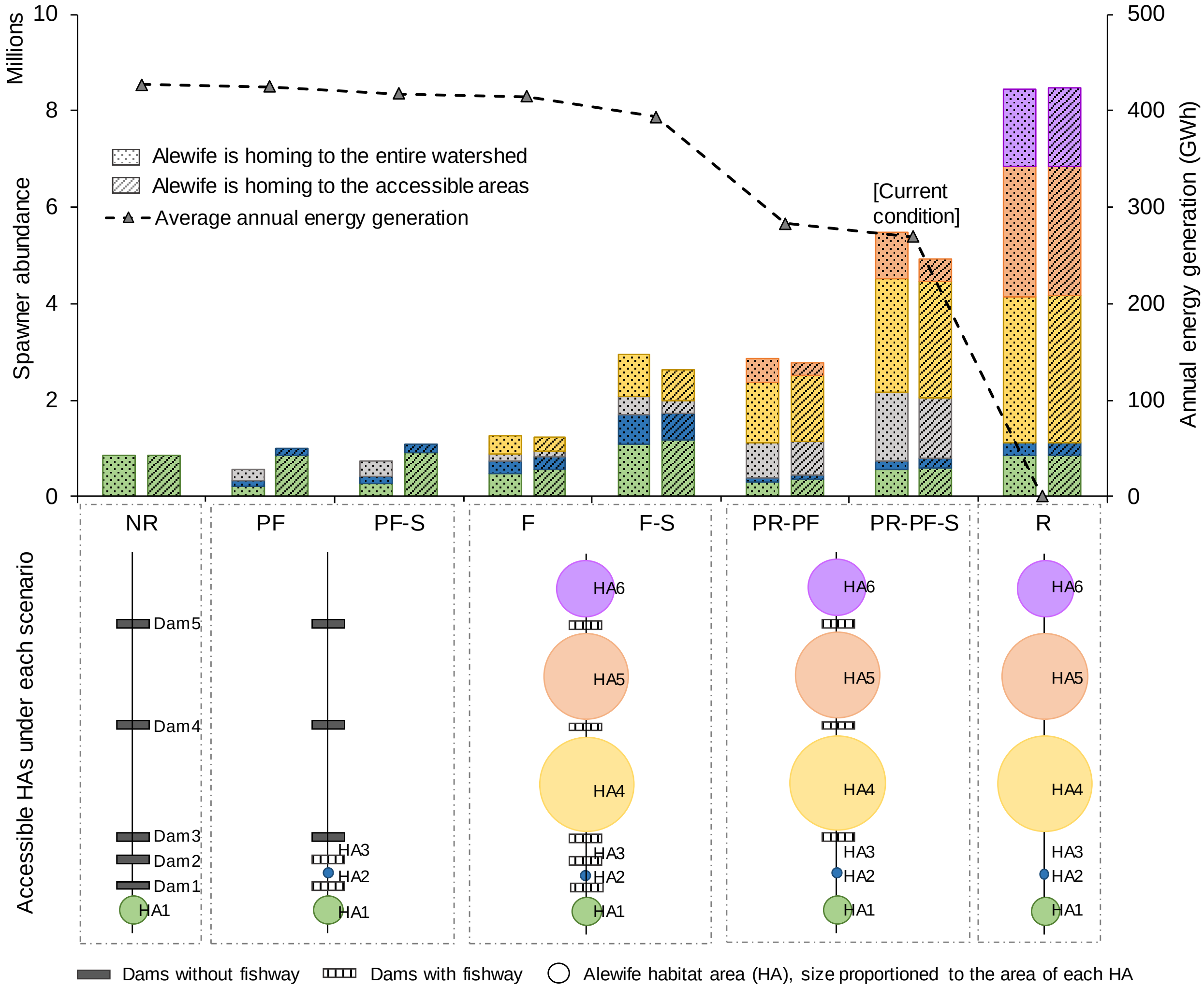


Figure 2-4. Tradeoffs between energy and Alewife spawner abundance under eight basin-scale dam management scenarios. Bars filled with the color of green, blue, grey, yellow, orange, and purple in the top portion are spawner abundance sequentially from HA1 to HA6 in the bottom portion. Stabilized spawner abundance of the two dispersal rules are shown as bars filled with dots (homing to the entire basin) and stripes (homing to the accessible areas).

2.3.2. Aggregated influence of upstream and downstream migration on fish population

Alewife spawner abundance was simulated for the two homing patterns, and results were very similar between the two. This further supports our previous conclusion that the different dispersal rules have limited effects on spawner abundance under the F scenario. Figure 2-5 illustrates the resulting population changes of alewife spawners homing to the accessible areas. Under a relatively low downstream passage rate of less than 60%, spawner abundance is lower than or similar to the NR scenario (the dashed line in Figure 2-5) and inversely related to the upstream passage rate. With this low downstream passage rate, reopening upstream habitat areas may have an adverse effect on the spawner abundance. This is because downstream mortality increases as improved upstream passage rates encourage more spawners to reach habitats upstream of one or more dams. Downstream passage is therefore a limiting factor for spawner abundance when it is 60% or less at each dam. Unless the downstream survival rate exceeds 60%, efforts or investments to improve upstream passage rates could be entirely ineffective. When downstream passage rates are relatively high (e.g., >70%), spawner abundance is larger than the NR scenario and positively related to both upstream and downstream passage rates. In this condition, the upstream passage rate becomes the primary limiting factor. When the upstream passage rate is lower than 60%, a 10% increase in downstream passage rate leads to less than 0.3 million increase in spawner abundance. However, when upstream passage rate surpasses 60%, spawner abundance is highly sensitive to changes in both upstream and downstream passage rates. A 10% increase in downstream passage rate can result in up to 2.7 million increase in spawner abundance. This shows a threshold exists related to the upstream passage rate, which needs to be accounted when designing dam management strategies. The upstream passage rate through a fishway has traditionally been used as a metric for assessing the success of restoration projects (Cooke and Hinch, 2013). However, our findings show that this is potentially misleading. Both upstream and downstream pass rates influence the objectives being considered when evaluating decisions related to dams (Pompeu et al., 2012).

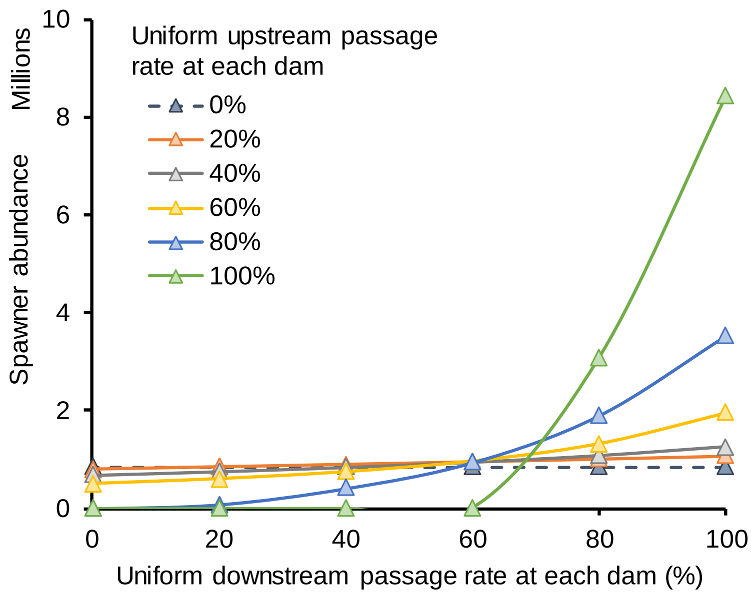


Figure 2-5. Alewife spawner abundance in the Penobscot River under various scenarios of upstream fishway passage rates and downstream passage rates. The colored lines correspond to various levels of upstream passage rates at all five dams.

2.3.3. Sensitivity analysis

Energy generation is sensitive to net head, turbine operation period, and overall efficiency regardless of the percentage of increase as these parameters have a linear relationship with energy (Equation 1). For spawner abundance, the absolute value of the sensitivity index in response to a −90% to −10% decrease and a 10% to 90% increase of model inputs are shown in Figure 2-6. Spawner abundance was the most sensitive to ocean mortality, spawning mortality, fishing mortality, the size of the habitat area, and the asymptotic recruitment level for all investigated ranges. The high sensitivity of alewife spawner to asymptotic recruitment level indicates the importance of increasing or maintaining a high habitat quality. In addition, spawner abundance was sensitive to any decrease, or less than 10% increase, in the alpha value and sex ratio. It was also sensitive to any decrease, or less than 70% increase in the fecundity slope. Accurate quantification of these sensitive variables is important in improving the confidence of model outputs.

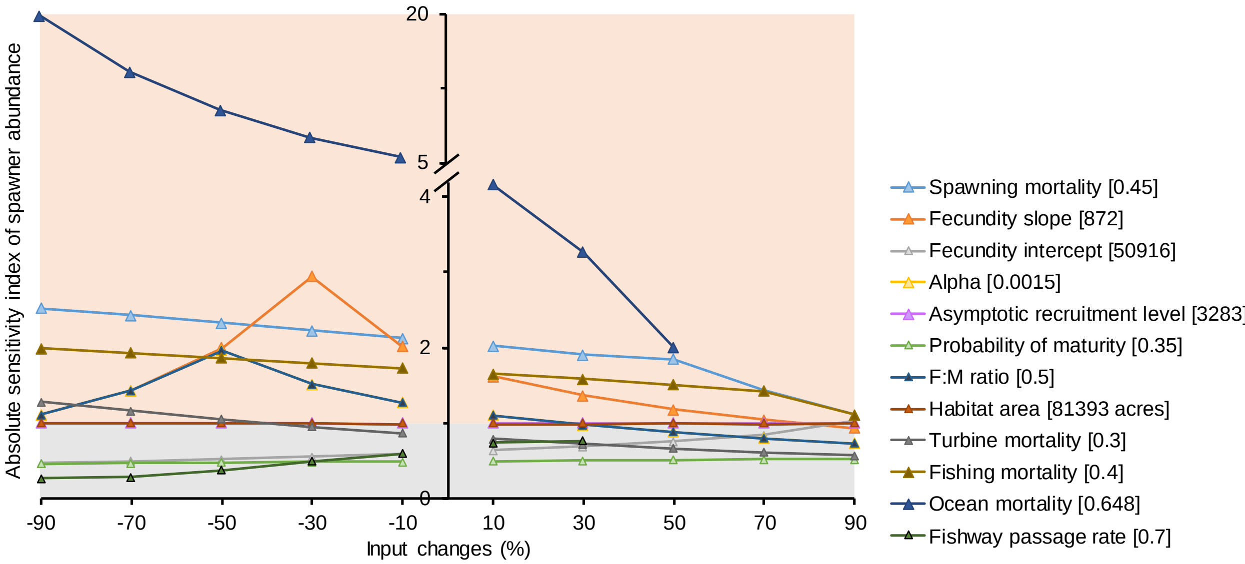


Figure 2-6. Sensitivity analysis index of alewife spawner abundance. Outputs of parameters distributed in the light orange shadow are considered highly sensitive, while those distributed in the light grey shadow are not. Numbers in the bracket represent the default value of each input parameter.

2.4. Policy Implications

As dam management decisions become increasingly contentious due to conflicting stakeholder interests, coordinated decisions that balance both energy production and fish abundance could be appealing (Roy et al., 2018). While dam removal is often heavily discussed and/or advocated when comes to dam decision-making, our results suggest that combining multiple dam management strategies including dam removals, fishway installations, and turbine shutdowns during the peak downstream migration periods could achieve a desirable fish restoration outcome, while preserving most of the hydropower capacity. Furthermore, the effectiveness of opening habitat through fishway installations is heavily influenced by the size of accessible upstream habitat and the downstream passage rates. For the Penobscot River, our analysis indicated that installing fishways in two lowermost dams could have minimal or even negative effect on alewife spawner abundance. This was mainly due to the unevenly distributed habitat areas in the watershed and potentially high cumulative downstream mortalities. This shows the importance of understanding the habitat distribution as well as upstream and downstream fish passage rates to inform proper decision-making associated with dam management. Our results also show that the commonly used "reopened/reconnected habitat area" could be an ineffective indicator of fish population recovery without an understanding of the potential upstream and downstream passage rates. Future studies also need to include all fish species for a comprehensive assessment of the energy-fish tradeoff.

While our study underscores the advantages of the systematic management actions made under the PRRP, such coordinated decisions are generally rare in the field (Opperman et al., 2011). One major barrier is the prevalence of private dam ownership, which can make basin-scale dam negotiations that involves multiple owners time and cost prohibiting. From a policy perspective, hydroelectric dams in the USA are licensed on an individual basis without a coherent basin-scale management plan, which reduces opportunities for co-optimization. Despite these significant challenges, there are a growing number of funding mechanisms and resources that encourage efficient basin-scale decisions (Owen and Apse, 2014). Compensatory mitigation is one funding model used to offset ecological damage caused by development in wetlands, and the US Army Corps of Engineers has established a method for including pro-environmental dam decisions in the compensatory mitigation scheme (USACE, 2008). Institutional initiatives and frameworks such as National Oceanic and Atmospheric Administration's Habitat Blueprint (Chabot et al., 2016) and US Department of Energy's Integrated Basin-Scale Opportunity Assessment Initiative reports (Kosnik, 2010a; Lowry, 2003) encourage basin-scale planning and there is growing federal support for this approach. Further research on the advantages of basin-scale dam decisions will support the use of these funding opportunities, improve co-optimization of fish and energy resources, and ultimately better reflect the preferences of stakeholders.

CHAPTER 3: BALANCING FISH-ENERGY-COST TRADEOFFS THROUGH STRATEGIC BASIN-WIDE DAM MANAGEMENT

3.1. Introduction

Energy generation, environmental impact, and cost are three major considerations influencing hydropower dam decision-making (Neeson et al., 2015; Opperman et al., 2011; Ziv et al., 2012). Depending on the type and context of dam management actions, tradeoffs among these three objectives often exist. One well characterized environmental impact of hydropower dams is the diminution of sea-run fish species (Brown et al., 2013; Limburg and Waldman, 2009). In response, fish conservation and restoration has become a required part of hydropower facilities' relicensing process under the regulations of the Federal Energy Regulatory Commission (FERC) (Emerson et al., 2012; Schramm et al., 2016). Hydropower operators are generally required to provide safe, timely, and effective fish passage. Efforts to mitigate these effects on migratory fish populations have included a wide range of engineered fish passage structures. Such structures are not guaranteed solutions and vary greatly in efficacy (Bunt et al., 2012; Noonan et al., 2012). More comprehensive improvement, such as dam removal may also be used to address impacts. All of these solutions usually require reductions in hydropower generation in order to accommodate operation (Kuby et al., 2005; Roy et al., 2018; Song and Mo, 2019; Song et al., 2019).

Operator responsibilities may also include safety issues associated with operation. In the US, over 60,000 dams will outlive their design lifespan by the late 2030s, posing a significant public safety risk if not repaired and maintained (O'Connor et al., 2015; USACE, 2016). Rehabilitation cost of the aged dams has been estimated to be a minimum of US$ 70 billion (Silva et al., 2019). Decision support that allows maximizing both hydropower generation and fish restoration, while minimizing cost is therefore imperative as not all dams are equal with regard to environmental impact or generating capacity.

The costliest dam management actions do not necessarily yield the best fish restoration or hydropower outcomes. In fact, such tradeoffs can vary significantly by river basin and by dam because the assemblage of dams can have synergistic influences on a river and its aquatic communities. To optimize these tradeoffs, numerous studies have noted the importance of basin scale or even multi-basin scale management as opposed to the traditional individual-based dam management (Neeson et al., 2015; Opperman et al., 2011; Roy et al., 2018) Fish-energy tradeoffs related to dams have been widely studied under diverse management options, including construction (Wild et al., 2018; Ziv et al., 2012), removal (Kuby et al., 2005; Null et al., 2014; Roy et al., 2018), fishway installation (Kuby et al., 2005; Song et al., 2019), or turbine shutdown (Eyler et al., 2016; Song et al., 2019; Trancart et al., 2013) at individual or basin scales. These studies highlight the advantages to managing dams at a larger scale but fall short of assessing the costs and operational efficacy for those that make the ultimate decision of what scale to work (e.g., the operators) and the decision-making incentives for management (in FERC).

Optimal solutions that balance fish-energy tradeoffs may be impractical when cost is considered. For example, fishway installation has been suggested as an effective way to balance fish-energy tradeoffs (Wild et al., 2018). However, the cost of a fishway may be twice as much as the average cost of dam removal (American Rivers, 1999; Strassman, 2011). Most previous tradeoff studies generally examine only a single type of management action. For example, Ziv et al. (2012) studied energy-fish-biodiversity tradeoffs under new dam construction scenarios in the Mekong River Basin. Null et al. (2014) analyzed tradeoffs between fish habitat gains and water supply losses under dam removal scenarios in California's Central Valley. Roy et al. (2018) also put emphasis on strategic dam removal and its influence on a wide array of tradeoffs at three watersheds in the New England region. To our knowledge, Song et al. (2019) is the only study that has investigated potential combinations of multiple dam management actions including dam removal, fishway installations, and turbine shutdowns for basin-scale dam management. The results of the study suggest that the optimal outcomes in hydropower generation and fish biomass may only be achieved when all three management actions are integrated. Therefore, a thorough investigation and analysis of fish-energy-cost tradeoffs associated with a full range of dam management options are pivotal to help support the making of sound and scientifically defensible decisions.

This study has three policy-relevant objectives. First, we detail a comprehensive analysis of fish-energy-cost tradeoffs under multiple dam management options, including dam removal and fishway installations on a basin scale. Second, we compare various dam management strategies using production possibility frontier curves to provide insights into the optimal strategies to balance energy-fish-cost tradeoffs. Third, we develop a dynamic modeling framework for basin-scale dam decision-making. This framework can be scaled and generalized to any region or river basin. It can also be used to facilitate dam negotiation process and engage stakeholders whose expertise and knowledge background may vary widely.

To achieve these objectives, a system dynamics model (SDM) was developed to simulate fish-energy-cost tradeoffs in dam decision-making. SDM is a computational method using a set of linked differential equations to dynamically simulate interactions within and among complex systems over a certain time period (Forrester, 1997; Sterman, 2001). It is a powerful tool to study multidisciplinary responses and tradeoffs of an action by capturing feedback loops and time delays among physical and biological components in a system (Cheng et al., 2018; Song et al., 2019). We use five hydropower dams located in the main stem of the Penobscot River, Maine to demonstrate the modeling framework. Three hypotheses were tested in this work. (1) There are dam management strategies that simultaneously maximize fish restoration potential and minimize hydropower loss and cost. (2) Basin-scale dam management strategies outperform individual dam management strategies in terms of maximizing energy and fish outcomes. (3) Diversifying dam management options can improve energy, fish, and cost outcomes in dam decision-making.

3.2. Materials and Methods

3.2.1. Proof of concept

The Penobscot River basin is a hotspot for both hydropower production and wild diadromous fish restoration. Hydropower in this basin alone accounts for around 22% of the total installed capacity in Maine (Kleinschmidt Group, 2015). These hydropower dams (as well as non-hydropower dams) have been implicated as the main reason for the substantial decline of native diadromous fish species (e.g., Atlantic salmon) with high commercial, ecosystem, and recreational values (NRC, 2002; Trinko Lake et al., 2012). To explore the fish-energy-cost tradeoffs associated with various dam management scenarios, we chose to study five hydropower dams located on the main-stem of the Penobscot River as a proof of concept. We note that two of the most downstream dams (Veazie and Great Works) have been removed in 2012 and 2013, respectively, as part of the Penobscot River Restoration Project (PRRP) (Opperman et al., 2011). The remaining three dams, from downstream to upstream, are the Milford Dam (with a Denil fishway and a fish lift), the West Enfield Dam (with a pool-and-weir fishway) and the Mattaceunk Dam (with a vertical slot fishway). This approach excludes several major tributaries of the Penobscot River and does not consider the complex fish passage paths near Marsh Island

(Stich et al., 2014). We note that the results from this research are intended to demonstrate the efficacy of such an approach, rather than being prescriptive for this watershed.

Our fish population modelling efforts were restricted to four of the twelve native diadromous fish species found in this system (Saunders et al., 2006) based on their high commercial, recreational, cultural, and ecological values: alewife (*Alosa pseudoharengus*), American shad (*Alosa sapidissima*), Atlantic salmon (*Salmo salar*), and sea lamprey (*Petromyzon marinus*). These four species may also undertake long distance migrations that historically distributed them throughout the reaches being modeled. Other species such as sturgeon (*Acipenser oxyrinchus* and *A. brevirostrum*), tomcod (*Microgadus tomcod*), smelt (*Osmerus mordax*) tend to exploit the lower river reaches making them less appropriate for this tradeoff simulation. Passage improvements for the catadromous eel (*Anguilla rostrata*) are less congruent with general fishway design and instead rely on climbing behaviors (Geffroy and Bardonnet, 2012; Jellyman, 1977; Watz et al., 2019). These fish also have a coastwide population structure (Jessop and Lee, 2016) making them less amenable to modeling within a single river system. Our four selected anadromous species spend most of their lives in the ocean to grow, but return to freshwater to spawn. Alewife (Barber et al., 2018), American shad (Bailey and Zydlewski, 2013), and Atlantic salmon (Fleming, 1998) have high rates of repeat spawning (iteroparity) over the course of their lifetime whereas sea lamprey spawn only once (semelparous) before death (Weaver et al., 2018).

Management scenarios. We assumed the baseline (worst) fish condition of each dam is complete obstruction of fish. Previous studies have shown that the effectiveness of fishways in facilitating fish upstream passage varies markedly based upon the types and numbers of fishway installed as well as the types of fish species (Bunt et al., 2012; Noonan et al., 2012). To capture these diversities, we simulated the installation of three widely adopted fishways: pool-and-weir fishway, Denil fishway, and fish lift (Table 3-1). It is not uncommon to have multiple fishways installed on a single dam. In this study, we assume up to two fishways can be installed on a dam simultaneously.

Therefore each dam has a total of eight potential management options (1) install pool-and-weir fishway, (2) install Denil fishway, (3) install fish lift, (4) install pool-and-weir and Denil fishways, (5) install pool-and-weir and fish lift, (6) install Denil and fish lift, (7) dam removal, and (8) no action. To provide a complete picture of the accumulated effects of multiple dams, we analyzed all possible permutations of the studied five dams (8^5 = 32,768 scenarios). When two fishways were installed, fish passage was assumed to be additive such that:

$P_{Total} = P_{Fishway1} + P_{Fishway2} * (1 - P_{Fishway1})$

representing the most optimistic outcome of using two structures.

Table 3-1. Description, passage rate, and capital cost of the studied fishways

Fishway	Description	Fishway upstream passage rate (%)								Capital cost per vertical meter ($ million/m)
		Alewife		American shad		Atlantic salmon		Sea lamprey		
		Mean	Ranges	Mean	Ranges	Mean	Ranges	Mean	Ranges	
Pool-and-weir	A series of small pools to create a long and slopping channel for fish to travel around the dam	36	6-73 (Bunt et al., 2012; Gahagan and Elzey, 2016; Nau et al., 2017; Sullivan, 2017)	19	0-70 (Beasley and Hightower, 2000; Bunt et al., 2012; Groux et al., 2017; Haro and Castro-Santos, 2012; Haro and Kynard, 1997; Sullivan, 2004)	48	0-100 (Bunt et al., 2012; Gowans et al., 2003; Holbrook et al., 2009; Lundqvist et al., 2008; Noonan et al., 2012)	18	1-35 (Castro-Santos et al., 2016; Haro and Kynard, 1997; O'Connor et al., 2003; Pereira et al., 2017)	0.178 (Nieminen et al., 2017)
Denil	A series of baffles with a relatively steep slope to reduce flow velocities	43	1-97 (Bunt et al., 2012; Haro et al., 1999; Nau et al., 2017; Stokesbury et al., 2015)	15	7-61 (Haro et al., 1999; Slatick, 1975)	76	12-100 (Holbrook et al., 2009; Noonan et al., 2012; Nyqvist et al., 2017a)	20	Passage efficiency estimate	0.190 (Nieminen et al., 2017)
Fish lift	An elevator to carry fish over a barrier	70	Passage efficiency estimate	35	6-67 (Groux et al., 2017; Larinier and Travade, 2002; Moser et al., 2000; Sprankle, 2005)	55	36-67 (Gowans et al., 2003; Noonan et al., 2012)	60	Passage efficiency estimate	0.237 (Porcher and Larinier, 2002)

3.2.2. Fish-energy-cost model

Six basin-scale objectives were chosen to evaluate candidate dam management scenarios: spawner population potential of four primary sea-run fish species (number of spawners), annual hydropower generation (GWh/year), and project cost ($ million). These measures were simulated using an integrated SDM model, consisting of age-structured fish population models, an energy model, and a cost model (Figure 3-1). SDM model was built in Vensim® DSS and runs on a daily time step.

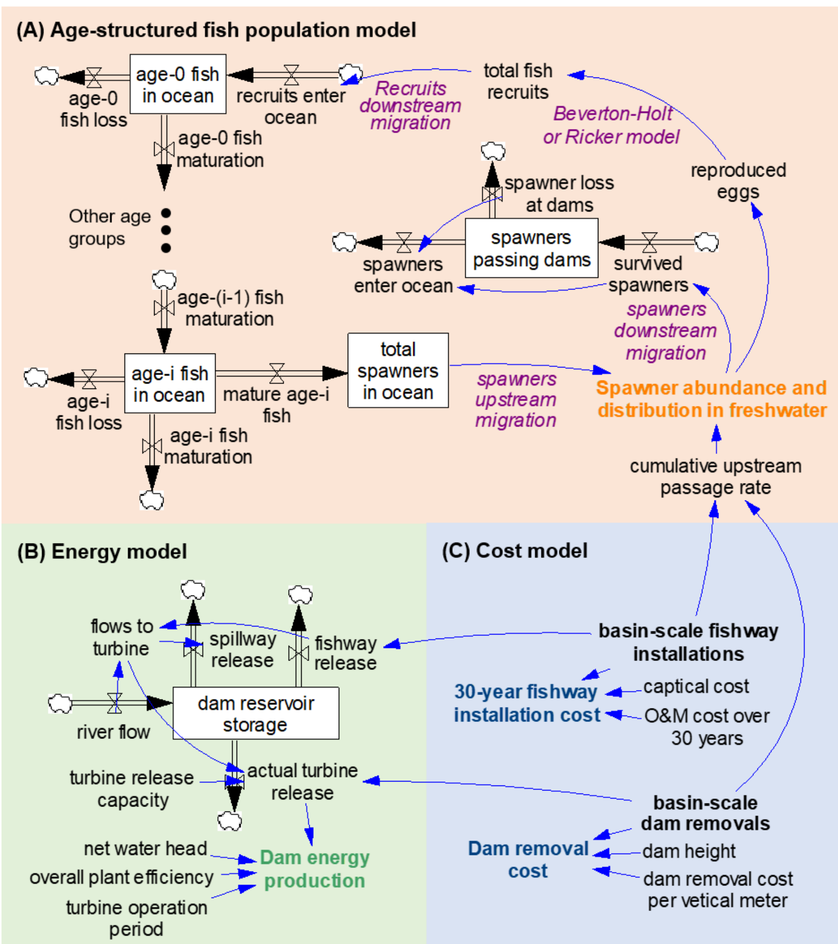


Figure 3-1. A simplified version of the integrated SDM model illustrating the key variables and connections of (A) age-structured fish population model, (B) energy model, and (C) cost model. The performances of fish population, hydropower generation, and cost are closely linked with dam management options on a basin scale.

The age-structured fish population model simulates spawner population potential of four fish species in the freshwater that are ready to spawn each year by keeping track of their growth, mortality, maturity, iteroparity, timing and period of migration at each life stage throughout the whole life span. The stabilized fish population potential was used in analysis by running the model 150 years. These models are modifications of four extant fish population models with alewife (Barber et al., 2018), American shad (Bailey and Zydlewski, 2013), Atlantic salmon (Nieland et al., 2015), and sea lamprey (Weaver et al., 2018). While the life histories of these species differ, we used a generalized format to account for the spawner potential under each scenario. The life cycle of each fish species starts from egg deposition in the freshwater, to recruit production in the freshwater, juvenile and post-spawners (excluding sea lamprey) seaward migration, the growth/maturity of fishes in the ocean, and spawning runs. Egg production each year was simulated as a product of the number of females that survived to spawn and their fecundity. Recruit production was determined by the carrying capacity of habitats and the spawner-recruit relationship. The Berverton-Holt spawner-recruit curve was adopted to simulate the recruit production of alewife (Barber et al., 2018) and Atlantic salmon (Nieland et al., 2013), while the Ricker spawner-recruit curve was used for American shad (Bailey and Zydlewski, 2013) and sea lamprey (Dawson and Jones, 2009).

For juvenile and post-spawn adults, seaward migration may require pass dams through spillways, turbine facilities, or fish bypass systems. The ratio of fish utilizing each route to pass a dam was assumed to be proportional to water being released through each route (Nyqvist et al., 2017b). Turbine mortality rate for all fish species was assumed to be 10% when passing each dam (Haro and Castro-Santos, 2012), while mortality rates of the other two migration routes were assumed to be zero as they are generally benign (Muir et al., 2001).

In the ocean, the number of fishes that can reach sexual maturity was determined by the ocean mortality rate and the probability of maturation (Table B1 of the Appendix B). Sexually mature females (i.e., spawners) swim to the freshwater to spawn. The number of spawners reaching a habitat area (HA_j) was determined by the cumulative upstream passage rate of dams downstream of HA_j as well as the dispersal rule described by Equations 3-1. We included the long-term blockage effect of dams that restricts fishes' motivation to seek habitats that were suitable for spawning but no longer accessible.

$$\begin{cases} S_{HA_j} = \left(\frac{A_j}{A} + \left(D_{HA_j} - \frac{A_j}{A}\right) \times \left(1 - P_j\right)\right) \times S, & \frac{A_j}{A} < D_{HA_j} \\ S_{HA_j} = D_{HA_j} \times S, & \frac{A_j}{A} \geq D_{HA_j} \end{cases} \quad \text{Equation 3-1}$$

where A_j and A are the size of habitat area j, HA_j and the total habitat area in the basin, respectively. j is a habitat area index which goes from 1 to 6, with 1 indicating the most downstream habitat area and 6 indicating the most upstream habitat area as segmented by dams. Habitat area sizes differ amongst the four fish species. The value of A_j and A for alewife and Atlantic salmon were obtained from (TNC, 2016) and (Nieland et al., 2015), respectively. The total habitat area, A, of American shad was calculated based upon Atlantic salmon total habitat area, assuming the ratio of the two is linearly proportional to the ratio between the two fish species' migration ranges within the Penobscot river basin (786 and 11,569 km for shad and salmon, respectively) (Trinko Lake et al., 2012). This is because both fish species have similar preference of free flowing river as their habitats (Greene et al., 2009; NMFS and USFWS, 2005). Once shad total habitat area was calculated, it was then allocated to the six river segments created by the five dams based upon the stream length of each segment to calculate HA_j (Trinko Lake et al., 2012). Sea lamprey habitat areas were assumed to be in the same size as Atlantic salmon's due to lack of field data as well as the similarity of preferred spawning habitat and migration range between the two species (Trinko Lake et al., 2012). P_j is the upstream passage rate of the dam located at the upstream of HA_j, the values of which are provided in Table 1. For a dam installed two fishways, the combined upstream passage rate, $P_{j,ab}$, was calculated based upon the passage rate of the two individual fishways, $P_{j,a}$ and $P_{j,b}$ using Equation 3-2. S_{HA_j} and S are the numbers of spawners in HA_j and the whole basin, respectively. D_{HA_j} is a dispersal factor calculated by Equation 3-3. D_{HA_1} equals 1.

$P_{j,ab} = P_{j,a} + \left(1 - P_{j,a}\right) \times P_{j,b}$ Equation 3-2

$D_{HA_j} = (D_{HA_{j-1}} - \frac{A_{j-1}}{A}) \times P_{j-1}$ Equation 3-3

The descriptions and governing equations of each life stage, as well as the value of input parameters were provided in Section B1 of the Appendix B. Particularly, this model captured the cumulative upstream and downstream impacts of all five dams on the distribution and population of spawners in the basin. Thus, it is capable to project relative changes in spawner population potential under various dam management alternatives.

The energy model simulates daily hydropower generation (MWh) by each of the five dams, which was calculated as a product of daily turbine release (m^3/s), net water head (meters), turbine operation period (hours), plant overall efficiency (assumed to be 0.85), water density (1000 kg/m^3), and gravitational acceleration (9.8 m/s^2) (Adeva Bustos et al., 2017; Hadjerioua et al., 2012; Singh and Singal, 2017). Daily stream flow data during the period of January 2001 to December 2015 at two nearby U.S. Geological Survey (USGS) stream gages (01034500 and 01034000) were used to estimate the river flows at the five studied dams using the drainage-area ratio method (Song et al., 2019). This 15-year data period was repeated 10 times for the modelled 150-year time horizon. Turbine release was determined by the relative values of three variables: river flow goes to turbine (the difference between river flow and flow demanded by fishway), the maximum turbine release capacity, and the minimum turbine release capacity (assumed to be 40% of the maximum capacity) (Table). Net water head of each dam was assumed to be its rated head obtained from (Amaral et al., 2012). Turbine operation period was assumed to be 24 hours per day. The energy model has been validated using a 15-year (January 2001 to December 2015) hydroelectricity dataset obtained from the U.S. Energy Information Administration (EIA, 2018; Song et al., 2019). Annual hydropower generation (GWh/y) was calculated as the average annual energy production over 15 years.

The cost model calculates total project costs related to fishway installation and dam removal. The revenue from hydropower generation was excluded due to its significant positive correlation with the energy generation estimated through the energy model. Fishway installation cost includes capital investment and operation and maintenance (O&M) cost over a 30-year planning horizon. This time period was chosen based upon the typical FERC license period for non-federal owned hydroelectric dams (Madani, 2011). Capital investment of fishway installation was estimated as a product of the dam height and the unit capital cost per vertical meter rise of the dam height (Table 1). Annual O&M cost was estimated to equal 2% of the capital cost of a particular fishway (Nieminen et al., 2017). Dam removal cost is a one-time investment which was simulated by multiplying the dam height with the average dam removal cost per vertical meter rise of the dam height ($ 0.173 million/meter) (Maclin and Sicchio, 1999).

3.2.3. Performance measures and calculations

Fish index is an indicator we created to represent the overall abundancy and diversity of the four fish species under consideration. The fish index was calculated using Equation 3-4.

$Fish\ index = \sum_{i=1}^{4} \frac{P_{ia}}{P_{im}}$ Equation 3-4 where i is a fish species index; P_{ia} is the spawner population potential of species i under a certain dam management alternative; P_{im} is the maximum spawner population potential of species i that the pristine river could support. We assume the value of P_{ia} under the scenario of removing all dams equals to the value of P_{im}. This approach administered equal value to each species.

Pearson correlation coefficients were used to quantity the correlations among various dam management options and the performance of the six basin-scale objectives described in Section 2.2. The Pearson correlation coefficients measure the linear association between two normally distributed random variables (Schober et al., 2018). It is a number between -1 and 1 that indicates the magnitude and direction of the association. A Pearson correlation coefficient between variable X and Y is calculated by Equation 3-5.

$$r = \frac{\sum_{i=1}^{n}(X_i-\overline{X})(Y_i-\overline{Y})}{\sqrt{\sum_{i=1}^{n}(X_i-\overline{X})}\sqrt{\sum_{i=1}^{n}(Y_i-\overline{Y})}} \quad \text{Equation 3-5}$$

The magnitude of the association for the absolute value of r was interpreted using Cohen's recommendation where 0~0.3 be interpreted as a weak correlation, 0.3~0.5 as a moderate correlation, and greater than 0.5 as a strong correlation (Cohen, 1988). The existence of a strong association does not imply a causal link between the variables.

The Pareto-optimal frontier defines the set of solutions for which none of the objectives can be improved in value by any other feasible solutions without worsening at least another objective value (Abbass et al., 2001; Almeida et al., 2019; Roy et al., 2018). To analyze tradeoffs between fish index, energy generation, and project cost under all dam management scenarios, we plotted the Pareto frontier with respect two out of the three criteria using the *geom_frontier()* function from the KraljicMatrix package in R.

3.2.4. Sensitivity analysis

We performed a Monte Carlo simulation for a dam management scenario that resembles the current condition of dam management in the Penobscot River (two dam removal and fish elevator construction) to understand the effects of parameters' uncertainties on spawner populations, hydropower generation, and project costs (Cheng et al., 2018; Sterman, 1984; Ventana, 2002). As installation of vertical slot fishway is not considered in this study, we assume the Mattaceunk dam has a pool-and-weir fishway given the similarities of the two fishways in fish passage performance and construction cost. The tested parameters, values, and ranges associated with fish population model and cost model can be found in the Section B2 of the Appendix B. Sensitivity analysis of the energy model was not carried out as we simulated hydropower generation is linearly related to associated variables (e.g., turbine release, net head, turbine operation period). The Monte Carlo simulation was repeated for 200 times.

3.3. Results

3.3.1. Fish- energy-cost tradeoffs of dam decision-making

The parallel coordinate plot in Figure 3-2 presents the key performance tradeoffs among the six objectives of interest: hydropower generation, project cost, and population potential of four primary sea-run fish species. Each vertical axis represents performance of the six objectives. The six objectives are oriented such that their performance improves moving vertically upward on each axis. Each polyline represents one of the 32,768 dam management scenarios and performance is designated by the points at which it intersects each vertical axis. The steepness of the diagonal lines between two adjacent axes displays the degree of conflict between the two objectives. The polylines are color-coded to represent the value of fish index which increases with colors changing from blue to red. The Pearson coefficient (r) among the six objectives as well as between the management options at each dam and the performance of six objectives at a 5% significance level was shown in Figure 3-3.

Energy and fish tradeoffs. Figure 3-2 shows a notable tradeoff between hydropower generation and the fish index, as only dark and light blue polylines (low fish index) occupy the top 20% of the energy axis while the dark red polylines (high fish index) are uniformly concentrated in the lower half of the energy axis. More specifically, preserving 95% of the installed capacity (405 GWh/y) accompanies 70~90% reduction of the fish index as compared to its maximum potential. On the other hand, preserving 95% of the fish index results in a 77% reduction of the installed hydropower generation capacity. Balanced management solutions can only be found where both energy and fish are around 60~66% of their maximum values, as indicated by the yellow polylines above 250 GWh/y of the energy axis. These balanced solutions are associated with removing any two of the three most downstream dams while installing at least one fishway at the remaining dams. On the other hand, certain dam management actions may result in both low energy generation and fish populations (e.g., blue polylines under 140 GWh/y). These outcomes mainly stem from management actions that only involve upstream dams while the most downstream dam(s) remains impassible. As shown in Figure 3-3, removing the two most upstream dams display moderate negative correlations with energy ($r = -0.6 \sim -0.5$) and negligible correlations with fish ($r \approx 0$).

Cost and fish tradeoffs. The dark red polylines (>80% of the maximum fish index value) are crowded in the area where project costs range from $9.3 to $23.6 million. The fish index increases with the increase of project cost until it reaches a threshold of nearly $24 million. Additional investment does not further increase fish index or even has an adverse effect on it. This is associated with management actions taken at upstream dams where the majority of fish population does not reach their immediate downstream habitat area (Song et al., 2019). This also occurs in management scenarios where fishway installation was chosen over dam removal. This is because fishway construction has a higher cost, but inferior performance in fish restoration, compared to dam removal (Magilligan et al., 2016; Nieminen et al., 2017). This explanation is demonstrated by the Pearson coefficients that indicate dam removals and fishway installations have a negligible negative ($r = -0.2 \sim -0.1$) and positive ($r = 0.1 \sim 0.4$) correlations with cost, respectively. In contrast, both options have positive correlations with the fish index ($r = 0.1 \sim 0.3$).

Energy and cost tradeoffs. Tradeoff between energy and cost is less substantial. The optimal solution in terms of both energy and cost is when all dams are preserved for power generation. Any other management actions tend to decrease energy and increase project costs as fishways are installed or dams are removed. The extent of decreased energy generation and increased project cost are closely related to the number of managed dams and the implemented options. In general, it is more cost effective to have fewer dams, further upstream with more generation capacity in terms of fish, cost, and energy management.

Fish-energy-cost tradeoffs. Project costs of dam management scenarios that simultaneously optimize fish and energy outcomes are in the range between $16.1 to $24.3 million (44% to 66% of the maximum cost). Only one of these scenarios comes with a project cost of lower than $17 million. This scenario involves removing the most downstream dam, installing Denil and fish lift fishway at the second dam, removing the third dam, install Denil fishway at the upstream two dams.

Tradeoffs among fish species. Relatively strong positive correlations ($r = 0.7 \sim 0.9$) present across the four fish species, except for the correlations between Atlantic salmon and American shad (r = 0.5) as well as Atlantic salmon and sea lamprey (r = 0.5). A lower correlation indicates potential conflicts in terms of restoration outcomes for different fish species. The three studied fishways, pool-and-wire fishway, Denil fishway, and fish lift, are considered effective in facilitating upstream passage of Atlantic salmon. However, American shad and sea lamprey may not effectively pass these fishways. Therefore, installing one or two of the three fishways may not simultaneously increase population potentials of all fish species. It is interesting to note that installation of the Denil fishway at the third dam can be negatively correlated with the population of American shad (r = −0.1). Furthermore, the installation of Denil or pool-and-weir fishway at the second or the third dam can be negatively correlated with the sea lamprey population potential (r = −0.1). This is linked to the low passage rates of the two fishways for American shad and sea lamprey as well as the severe turbine kills when post-spawn adults and juveniles migrate downstream. In this condition, fishways may work as ecological traps and potentially cause a further collapse of the regional fishery (Pelicice and Agostinho, 2008).

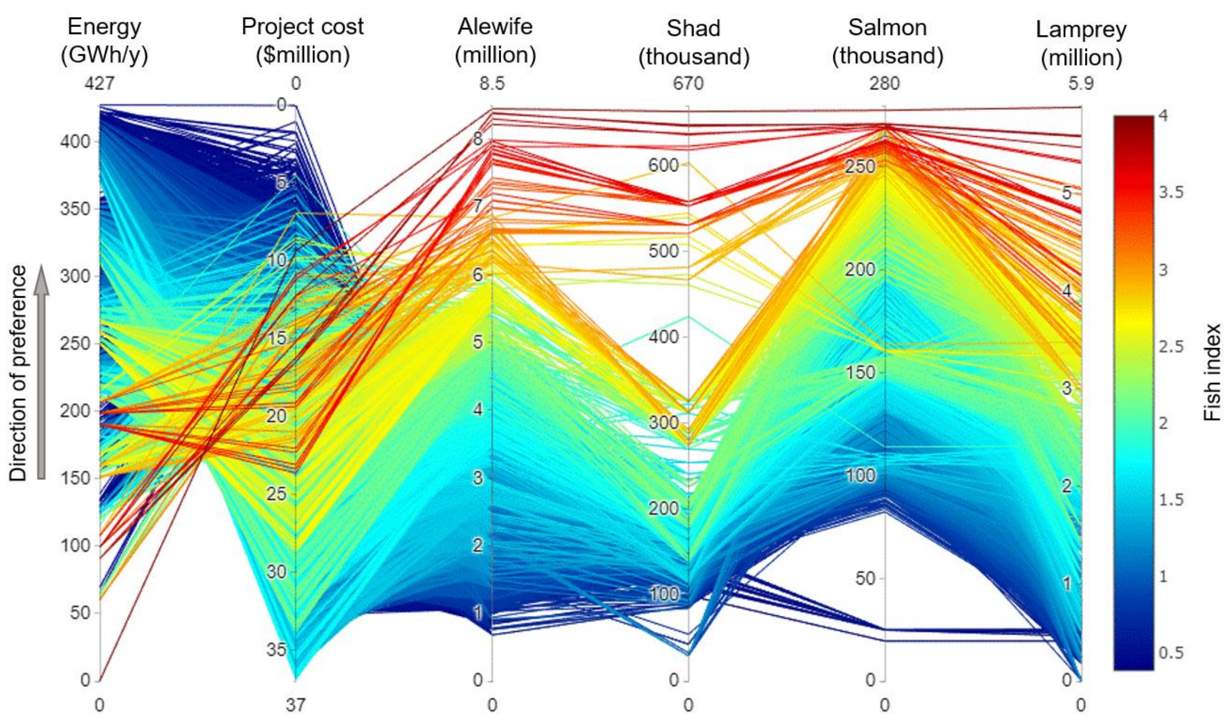


Figure 3-2. Parallel coordinate plot of tradeoffs among seven objectives for all basin-scale dam management scenarios in the Penobscot River. Each y-axis indicates one objective. The arrow indicates the preferred direction of all objectives. Each polyline is one dam management scenario which color-coded by the value of fish index.

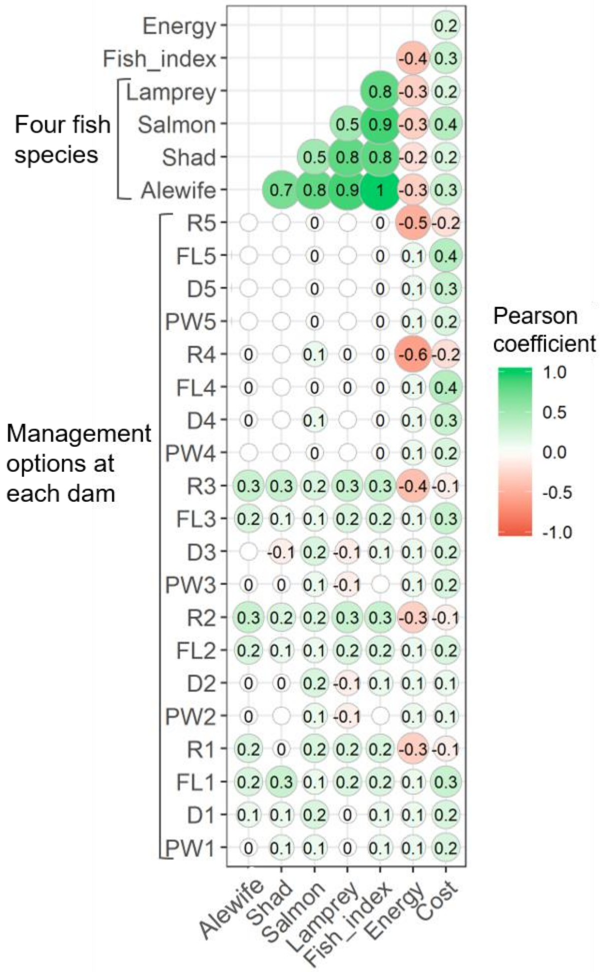


Figure 3-3. Pearson coefficient among management options at each dam and the performance of six objectives at the 0.05 level. Some of the cells are blank, meaning that the correlation detected is not considered to be significant. PW stands for pool-and-weir fishway, D for Denil fishway, FL for fish lift, and R for removal. The number following these initials refers to the studied five dams, among these 1 to 5 refer to dams from downstream to upstream: the Veazie, Great Works, Milford, West Enfield, and Mattaceunk Dam.

3.3.2. The effectiveness of dam management strategies

Given the dense nature of cloud of potential dam management scenarios showing in parallel coordinate plot (Figure 3-2), we projected the performances of fish index, energy, and project cost onto two-dimensional scatter plots to get further insight into their inherent tradeoffs as well as to determine the Pareto-optimal frontier.

Individual vs. basin-scale dam management strategies

Figure 3-4 is a comparison of the fish-energy tradeoff performances between individual dam management strategies (scenarios that only include management action at one out of the five dams) and basin-scale strategies (scenarios that have management action at least two dams). While the individual dam management strategies are likely to preserve a high percentage of the hydropower generation capacity, our results show that basin-scale management strategies can significantly improve fish index while preserving a similar amount of hydropower generation capacity. This shows that the basin-scale management can more effectively balance fish-energy tradeoffs than individual management as our second hypothesis stated. It also indicates the importance of strategically managing dams on a basin scale to achieve balanced outcomes between two competitive interests. For individual dam management strategies, scenarios that lead to increase of fish index are associated with managing the most downstream dam. This finding highlights the importance of prioritizing the enhancement of fish passage performance of the most downstream dam to recover migratory fish species.

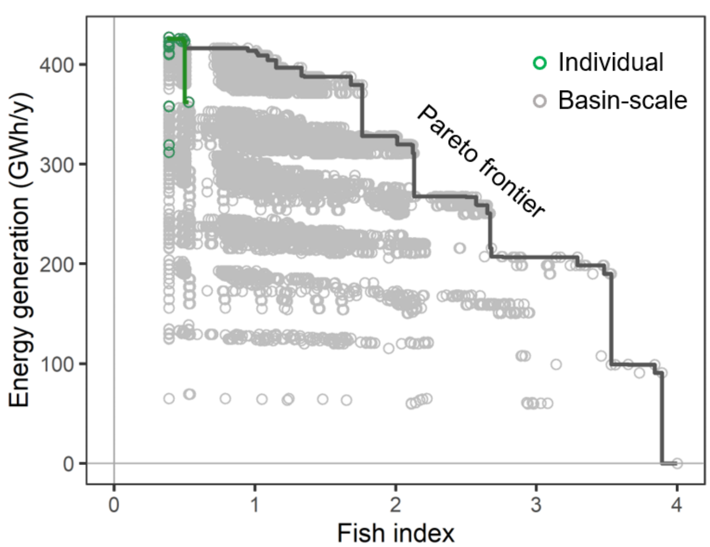


Figure 3-4. Fish-energy tradeoffs under individual (green circles) and basin-scale (grey circles) dam management scenarios. Each point corresponds to a polyline in Figure 3-2.

Single vs. diversified management options

The impacts of single and diversified management options for basin-scale dam management were analyzed by dividing all scenarios into four groups: (1) no action for all five dams, (2) only implement fishway installations, (3) only implement removals, (4) integrate fishway installations and removals. For fishway installations, we further separated them into two groups: management options with only 1 fishway allowed at each dam versus management options with up to 2 fishways allowed at one dam.

From a fish-energy perspective (Figure 3-5 (a)), management scenarios that only involve fishway installations (yellow and orange circles) are mostly effective in terms of preserving hydropower generation capacity. However, they have limited benefit in terms of fish restoration. This is because of the relatively low upstream passage performance for fish that need to ascend sequential dams, even though each dam has a high upstream passage rate (Song et al., 2019; Sweka et al., 2014; Winemiller et al., 2016). For example, only 33% spawners can reach to their spawning habitat areas located on the upstream of five dams even if each dam's upstream passage rate is 80% (relatively high). It is notable that none of the management scenarios that only involve dam removals (red circles) reside on the Pareto frontier curve. This indicates that dam removal alone cannot optimize both energy generation and fish restoration. On the other hand, dam management scenarios that integrate fishway installations and dam removals (light and dark blue circles) occupies the majority of the "turning point" of the Pareto frontier curve, indicating optimal solutions simultaneously maximize energy and fish populations. From a fish-cost perspective (Figure 3-5 (b)), the Pareto-efficient scenarios are those with the least cost at each level of fish index, which are mainly management scenarios that integrate fishway installations and removals. From an energy-cost perspective (Figure 3-5 (c)), although management scenarios that only involve dam removals are the cheapest solutions at each level of energy, management scenarios that integrate fishway installations and removals can generate similar energy with a slightly higher cost. Taking all three aspects into consideration, we conclude that diversifying dam management options have the highest potential in balancing fish-energy-cost tradeoffs.

Allowing multiple fishways to be installed on a single dam also has a significant effect on the fish-energy-cost outcomes. For management scenarios that only involve fishway installations, allowing installation of two fishways on each dam can increase the possibility of improved fish index up to a value of 1.8, while preserving a similar amount of hydropower generation capacity. However, this comes at a cost of higher project investment. This is because the performance of fishways typically differs among species (Noonan et al., 2012). For example, pool-and-weir and Denil fishways have high passage rates for Atlantic salmon but low passage rates for American shad, alewife, and sea lamprey (Table 3-1). Fish lifts generally perform well for the upstream passage of most fish species. Therefore, installation of multiple fishways at one dam may facilitate upstream migration of multiple species. Similarly, for the scenarios that integrate fishway installations and dam removals, allowing installation of multiple fishways on a single dam can also markedly increase the value of fish index while preserving a similar amount of energy as compared with scenarios that only install one fishway. These findings further confirm that diversifying dam management options by allowing tailored fishway design and installations targeting multiple fish species can further benefit the optimization of the fish-energy-cost outcomes.

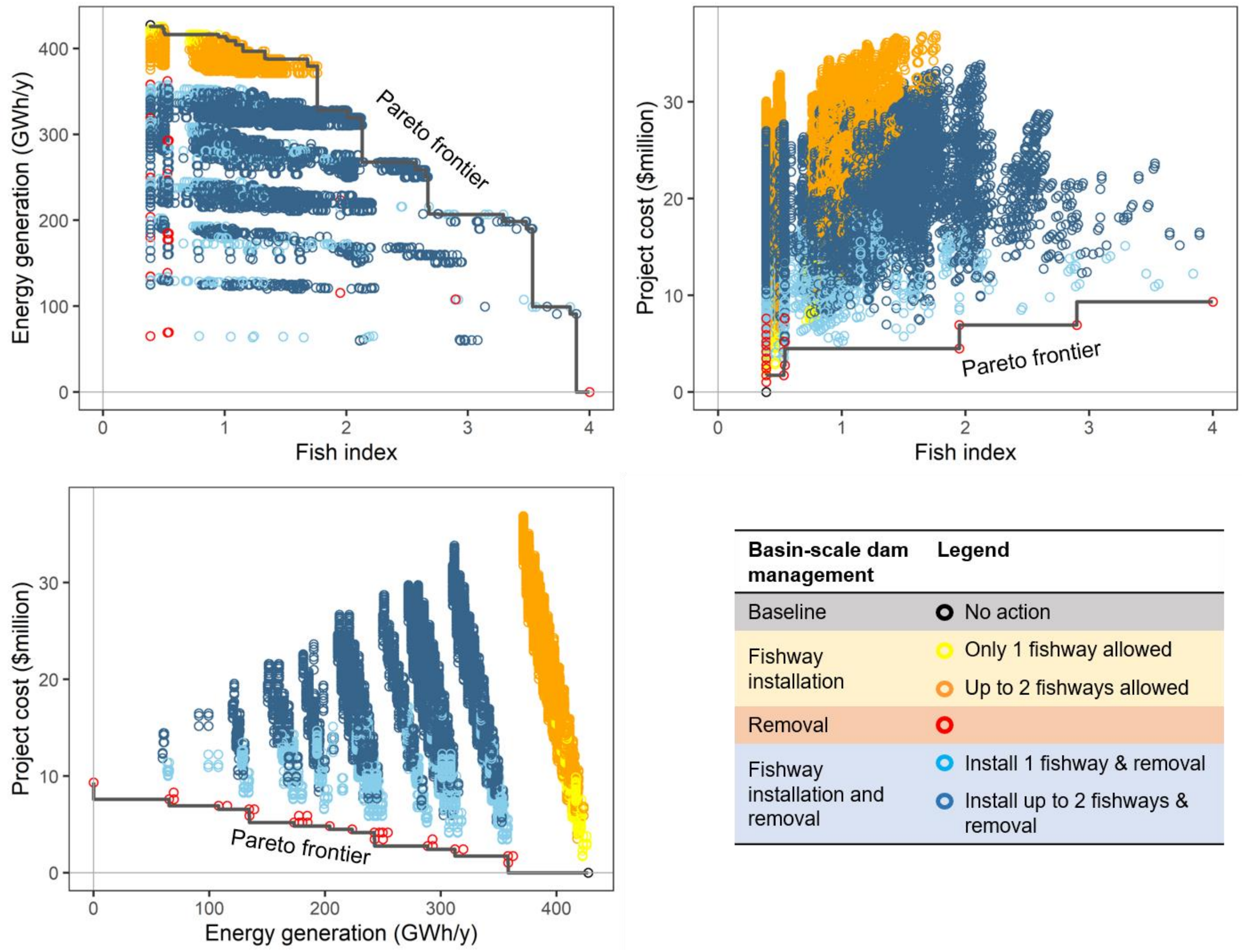


Figure 3-5. Tradeoffs among fish index, energy generation, and project costs for all basin-scale dam management scenarios of five main-stem dams in the Penobscot River basin. Each point represents a basin-scale dam management scenario.

3.3.3. Sensitivity analysis

Figure 3-6 presents trajectories of fish spawner population potential associated with 50%, 75%, 95%, and 100% likelihood for the tested scenario in response to changes of input parameters. Spawner population potentials of alewife, American shad, Atlantic salmon were found to change at a range of 0~13.9 million, 0~0.9 million, and 42~386 thousand, respectively, with a 100% confidence in the studied river basin. The results also show that spawner population potential of these three species reach equilibrium under all scenarios. This phenomenon can be explained as an outcome of the necessary biological process of density dependence, usually in early life history (Quinn and Collie, 2005). It has to be noted that these equilibriums are a result of the simplified mathematical assumptions for testing theoretical sensitivities, while there are numerous uncertain and stochastic factors can result in population potential variations in reality. The equilibrium of sea lamprey spawner population potential is sensitive to parameters' uncertainty. With a 50% confidence, sea lamprey spawner population potential stabilize at a range of 0~7million. Otherwise, it presents a regular oscillation every 9 years which is consistent with one life cycle of sea lamprey. This is a mathematical result of the Ricker curve, a density dependent spawner-recruit curve presented (Myers, 2001). This curve determines that recruits of sea lamprey are reduced at high spawner population levels and increase at low population levels. For project cost, it is in the range of $7.4~$34.4 million with a 100% confidence.

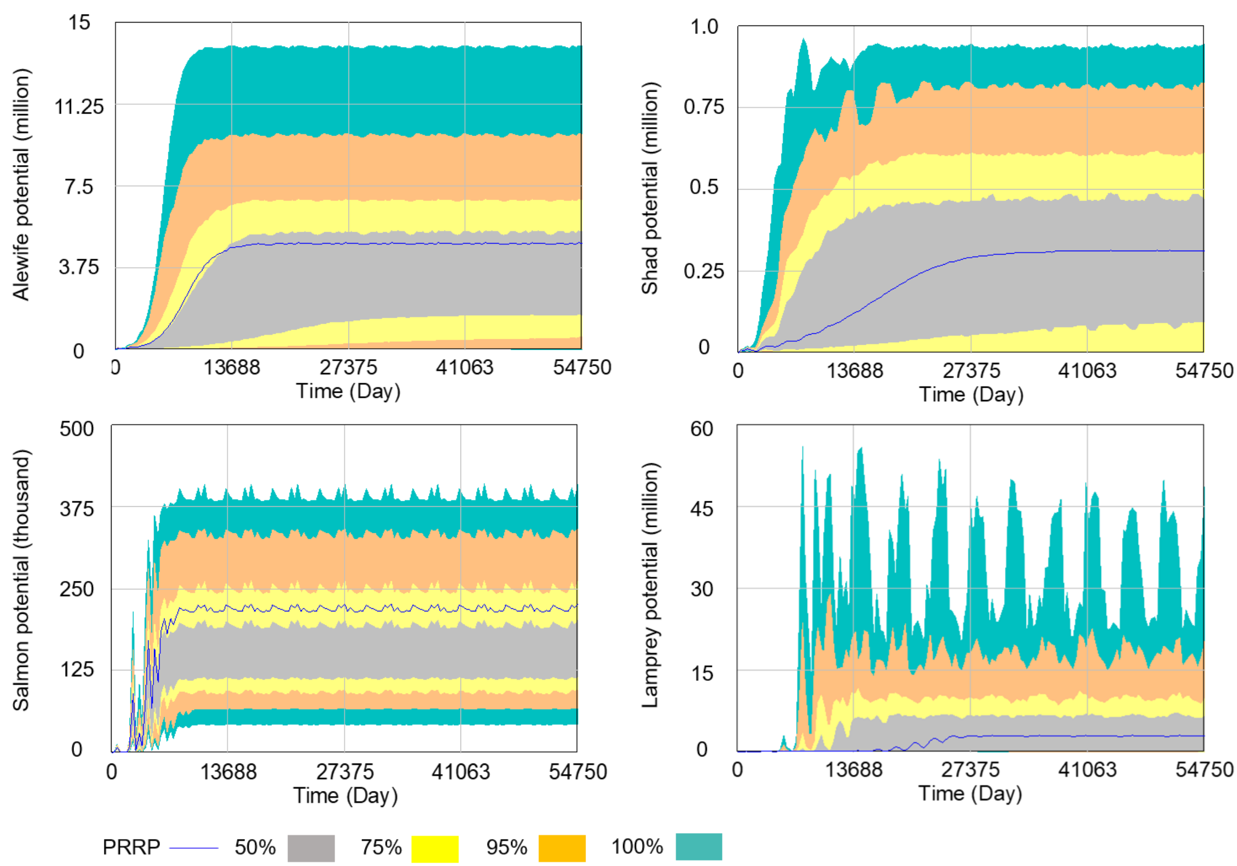


Figure 3-6. Monte Carlo simulation of the age-structured fish population model, presenting trajectory spawner population potential of (a) alewife, (b) American shad, (c) Atlantic salmon, and (d) sea lamprey in response to changes of input parameters.

3.4. Conclusion and Policy Implications

This dynamic modeling framework utilizing the system dynamics modeling technique was developed in this study to examine various dam management options. Using the Penobscot River as a testbed, it was found that it is possible to maximize fish potential and hydropower generation to 60-62% of their highest achievable values while limiting the project cost to $17 million (44% of the highest possible project cost). Our results also show that basin-scale management strategies can significantly improve fish index while preserving a similar amount of hydropower generation capacity as compared to management strategies that only focus on individual dams.

This integrated basin scale approach we describe is distinct from the current practice where dam decisions are often made in isolation and are primarily based upon the interests of the individual dam owners (Graf, 2001; Moran et al., 2018). Our results clearly demonstrate the advantage of dam management at a basin-scale for simultaneously optimizing energy, fish, and cost outcomes. This further highlights the importance of engaging a broad range of stakeholders who can influenced by dam decisions, especially those that have been rarely engaged in the decision-making process (Fearnside, 2015; Siciliano et al., 2015). Incorporating stakeholder inputs in the FERC hydropower relicensing process could be an important initial step in achieving this goal. When the dam management is done from a basin scale, diversification of management options (e.g., combination of fishway installations and dam removals) as well as implementation of fishways targeting multiple fish species can better balance fish-energy-cost tradeoffs.

The modeling framework developed in this study may be extended spatially and temporally to other river basins to address specific real-world challenges. Such an approach is not intended to make a decision, but rather to inform those upon whom that responsibility rests. Specifically, these models can be used to facilitate the discussions among stakeholders and decision-makers for consensus building in pursuit of the best possible economic, environmental, and social outcomes. Real-world decision-making may involve more criteria than those that have been considered in this study, such as flood control, recreation, water supply, sediment contamination/accumulation. Future work may involve incorporation of additional criteria that might be of interest to the stakeholders and decision makers.

CHAPTER 4: A TEMPORAL PERSPECTIVE TO DAM MANAGMENET: INFLUENCE OF DAM LIFE AND THRESHOLD FISHERY CONDITIONS ON THE ENERGY-FISH TRADEOFF

4.1. Introduction

Balancing hydropower generation and fish population to meet both human and ecosystem needs has become a pressing issue of dam decision-making (Grumbine and Xu, 2011; Poff and Olden, 2017; Wild et al., 2018; Winemiller et al., 2016; Ziv et al., 2012). There are more than 45,000 large dams (>15 meters in height) around the world, which are mainly used for hydropower generation and irrigation (Bartle, 2002; Vörösmarty et al., 2010; WCD, 2000). In addition, over 3,700 large hydroelectric dams with a total capacity of more than 1,000 GW are to be constructed in the next few decades, which will increase current hydropower generation by more than 70% (Zarfl et al., 2015). Though these dams play a key role in meeting the increasing energy demand, they pose a great risk to sustainable fisheries (Limburg and Waldman, 2009; Song et al., 2019) as well as the wellbeing of fish-dependent communities (Chen et al., 2016; Limburg and Waldman, 2009; Song et al., 2018; Winemiller et al., 2016; Zarfl et al., 2015). Dams can substantially decrease fish populations by fragmenting migration corridors (Beasley and Hightower, 2000; Hall et al., 2011), degrading habitat quality (e.g., changes in temperature and discharge) (Johnson et al., 2007; Piffady et al., 2013; Zhao et al., 2012), and causing severe turbine injuries (Schaller et al., 2013; Stich et al., 2015). In the North Atlantic basin across US, Canada, and Europe, the abundance of 23 out of 24 diadromous fish species has dropped to less than 10% of their historical levels as a result of heavy dam construction (Limburg and Waldman, 2009). Some of these species (e.g., Atlantic salmon) are currently listed as endangered species under the federal Endangered Species Act (Lichter et al., 2006).

Such energy-fish nexus has manifested in many previous, current, and future dam decisions. For example, Mekong River is currently under an ambitious agenda of hydropower development. Eleven hydroelectric dams have been proposed to be constructed on the lower main stem across Laos, Thailand, and Cambodia (Grumbine and Xu, 2011; ICEM, 2009). Once completed, these dams will generate roughly 15,000 MW of hydropower, projected to account for 8% of the regional demand by 2025 with $3.7 billion/year of gross income (Grumbine and Xu, 2011). However, it has been estimated that these projects would reduce up to 30% of annual protein intake by the national populations of Laos and Cambodia (Grumbine and Xu, 2011). Similar energy-fish tradeoff studies under various new dam construction or dam removal scenarios have been performed in other regions, including the entire Mekong River basin (Ziv et al., 2012), the Willamette basin (US) (Kuby et al., 2005), the Penobscot River basin, (US) (Song et al., 2019), New England watersheds (Roy et al., 2018), and the Oir River basin (France) (Trancart et al., 2013). Researchers found that desired energy-fish outcomes may be achieved by strategically removing or avoiding building dams at locations that significantly affect fish migration (Kuby et al., 2005; Roy et al., 2018; Song et al., 2019; Ziv et al., 2012). Other studies have also found that installing effective fish upstream passage structures (hereafter referred to as fishways) (Katopodis and Williams, 2012; Larinier, 2000; Null et al., 2014; Thorncraft and Harris, 2000) and properly shutting down turbines during fish peak downstream migration period (Eyler et al., 2016; Trancart et al., 2013; Watene and Boubée, 2005) can be effective ways in balancing the energy-fish tradeoffs.

However, previous studies on dam related energy-fish tradeoffs have widely used simplified proxies, such as habitat gains (Null et al., 2014; Roy et al., 2018) and reconnected areas (Kuby et al., 2005; Ziv et al., 2012), to estimate the potential changes in fish populations. These indicators are fixed values which do not reflect the temporal changes of fish populations in response to different dam management activities. The temporal perspective provides important information regarding whether the effects take place relatively rapidly or slowly, potential time delays in response to certain management actions (Limburg and Waldman, 2009), as well as the key temporal thresholds for a certain phenomenon to occur (e.g., depletion of a fish stock) (Rodríguez et al., 2006). Such information is fundamental to inform the type of dam management efforts needed and the best timing of conducting these efforts. However, only a few previous studies have examined the temporal changes of fish populations in response to dam management actions. Burroughs et al (2010) measured the response of fish communities to the removal of Stronach Dam, a 2-MW hydroelectric dam on the Pine River, Michigan. They found that the abundance of Brown trout and rainbow trout increased by more than twofold 4 years after the dam removal (Burroughs et al., 2010). Lundqvist et al (2008) predicted temporal changes of salmon populations passing a fish ladder during a 20-year period using a matrix population model. They found that a fivefold population increase in 10 years can be achieved by improving fishway upstream passage rate from the current 30% to around 75% (Lundqvist et al., 2008). Nevertheless, none of the previous studies investigated the influence of a dam's life span on the rate of fish population decline and the time needed for fish recovery once the dam is removed. Furthermore, none of the studies have further linked the dynamic fish population changes to the losses/increases of hydropower generation to explore the temporal energy-fish tradeoffs.

To address these knowledge gaps, this study aims to answer the following question: How do dam life span, upstream passage rate, and downstream habitat area influence the temporal changes of fish population, fish restoration period, and the energy-fish tradeoff? In order to achieve this goal, we chose system dynamics model (SDM) to simulate the temporal energy-fish tradeoffs under different dam life span and management scenarios. SDM is a computational method that simulates the behavior of different components of a complex system over a certain time period. Particularly, SDM can capture the embedded feedback and interactions among different system elements using a set of linked differential equations (Forrester, 1997). It is an appropriate approach to simulate the dynamic energy-fish changes which has been previously applied in modeling temporal hydropower production (Bosona and Gebresenbet, 2010; Sharifi et al., 2013) and fish abundance (Barber et al., 2018; Ford, 2000; Stich et al., 2018) separately. However, the impacts of dam management on the energy-fish tradeoffs, especially those from a temporal perspective, have not been studied yet. This study adapted an energy-fish model presented in (Song et al., 2019) to explore temporal dam management strategies. Case selection, methodologies, and data sources are described in section 2. Section 3 conducts results analysis and discussion. Section 4 is the conclusions, significance, and policy implications of this study.

4.2. Materials and Methods

4.2.1. Study site

The Penobscot River is the second largest river system in the New England region of the US with a drainage area of 22,300 km^2 (NRCM, 2019). The river system historically provided important freshwater habitats for 12 native sea-run fish species (Schmitt, 2017). However, these fish populations have declined significantly after a heavy damming period between late 1800s and early 1900s. The Milford Dam is currently the lowermost dam on the main stem of the

Penobscot River, around 61 kilometers away from the river mouth (Figure 4-1) (Maynard et al., 2017). It is a run-of-river hydroelectric dam with an installed capacity of 8 MW (Amaral et al., 2012). This modeling exercise will be focused on the Milford Dam, given that it is the first anthropogenic barrier to a vast amount of upstream habitat areas. The fishway performance in facilitating fish upstream migration at this site is critical in determining the distribution and abundance of native diadromous fish species (Gardner et al., 2012; Gardner et al., 2013; Gehrke, P. et al., 2002; Tonra et al., 2015). Decision-making of this dam also presents interesting energy-fish tradeoffs. We modelled alewife (*Alosa pseudoharengus*) as a representative sea-run fish species because it is ecologically important in freshwater, estuarine, and marine environments, providing food source for other species such as brown trout, Atlantic cod, and other aquatic furbearing mammals (ASMFC, 2009; Dalton et al., 2009; McClenachan et al., 2015). In addition, alewife is at the focal point of restoration as their commercial harvest has dropped from around 3.5 million pounds in the 1950s to less than one million pounds as of 2000 (Goode, 2006; MaineDMR, 2018). This dramatic decline was considered to be attributed to existing dams.

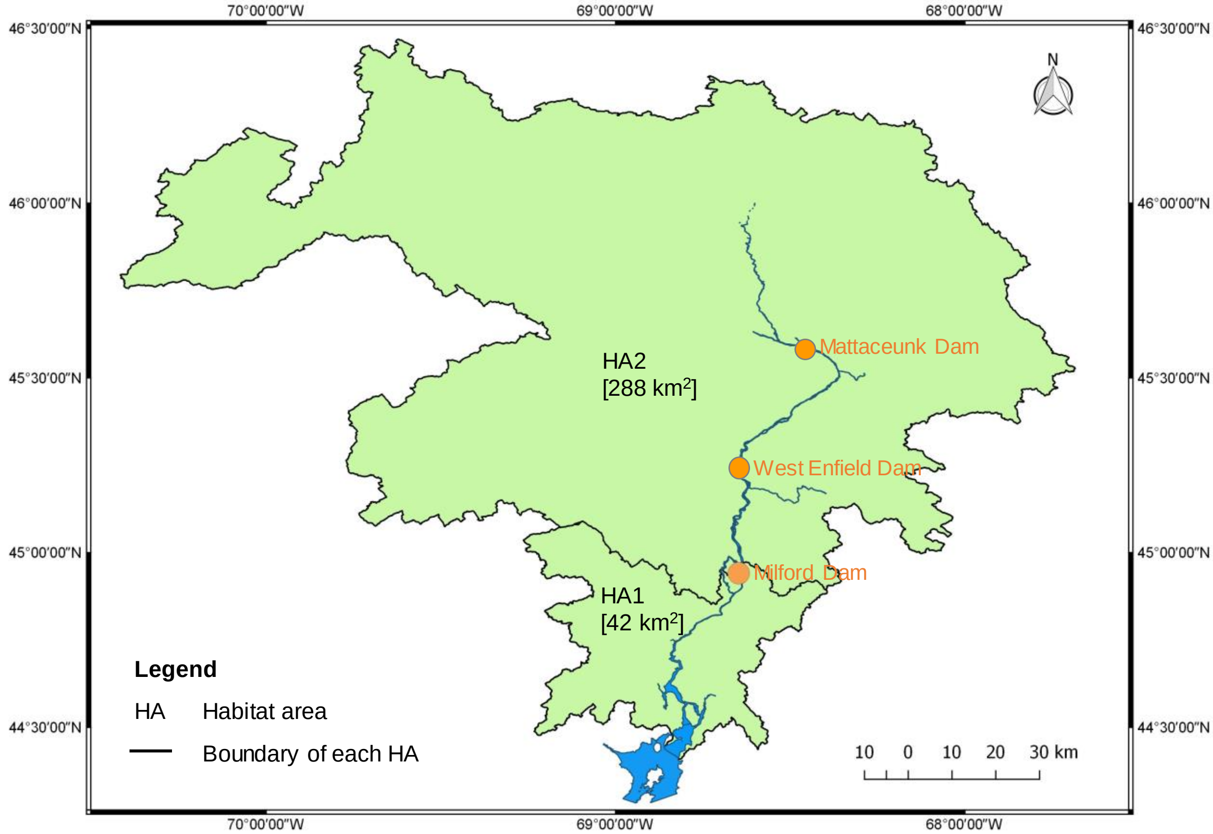


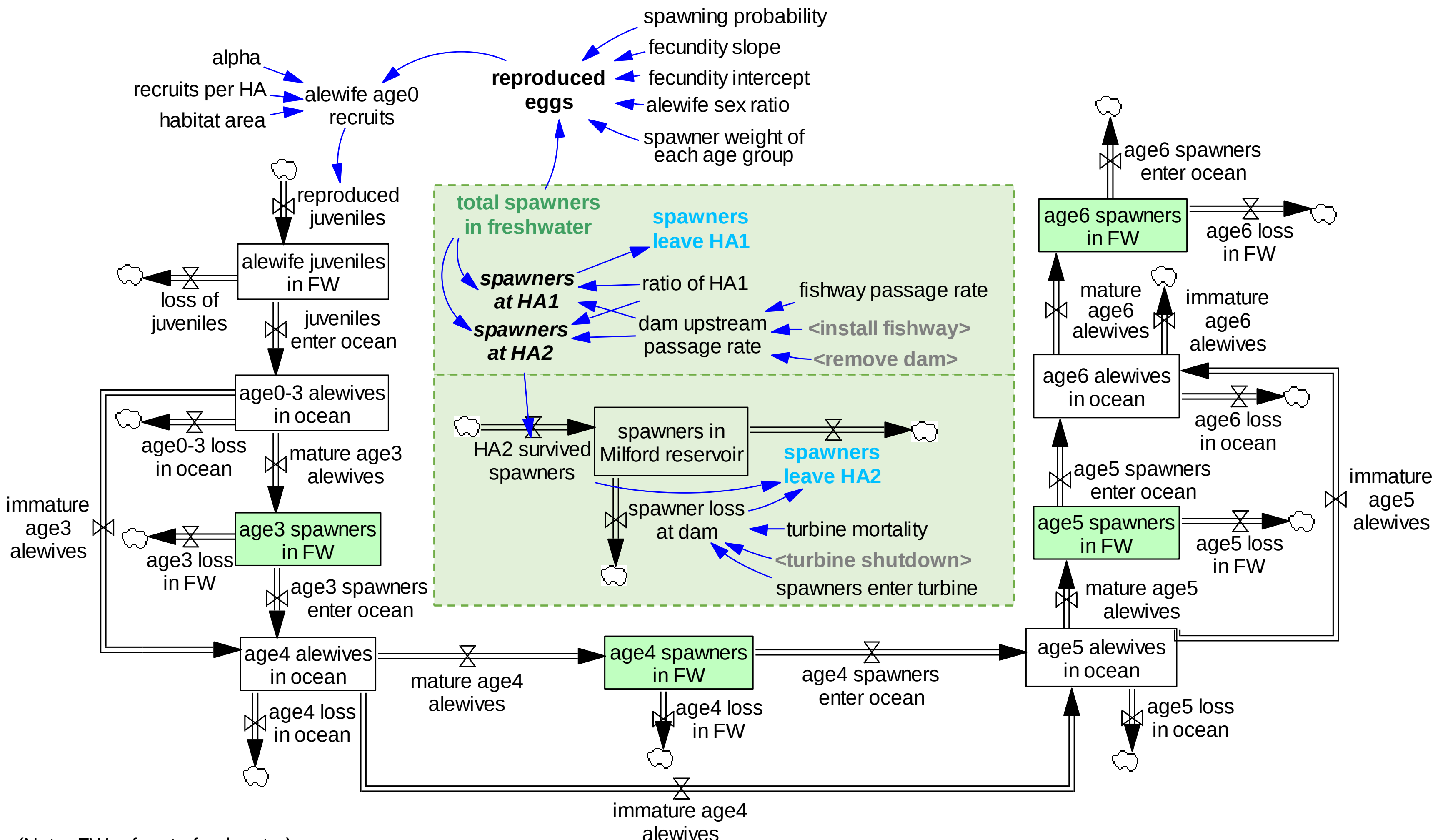
Figure 4-1. Map of the study area showing the locations of Milford Dam as well as the size of alewife spawning lakes/ponds in the Penobscot River Basin.

4.2.2. System dynamics modeling of the energy-fish nexus

A quantitative SDM usually consists of four elements: stocks, flows, auxiliary variables, and connectors (Ford, 2000). Stocks are variables that accumulate or deplete over time (e.g., populations of different alewife age groups). Flows represent the inflows and outflows of a stock (e.g., maturation or death in an alewife age group), which determine the stock's rate of change. Auxiliary variables are other important endogenous and exogenous variables that influence system behavior. Connectors show the flow of information in the system and links the stocks, flows, and auxiliary variables. The connections among the four elements are usually visualized as stock-and-flow diagrams. In this study, we used Vensim® DSS to develop the stock-and-flow diagrams and the energy-fish model.

The energy-fish model was built upon an existing model of the Penobscot River basin developed in (Song et al., 2019). The Song et al. (2019) model has been validated using the historical alewife landing and hydropower production data obtained from the Department of Marine Resources (MaineDMR, 2018) and the U.S. Energy Information Administration (EIA, 2018b), respectively. This validated model was adjusted in this study to include the three existing dams on the Penobscot main stem: Milford dam, West Enfield dam, and Mattaceunk dam (Figure 4-1). For simplicity, we assumed the upstream pass rates of the West Enfield dam and the Mattaceunk dam to be 100%, given that they each have installed at least two types of fishways with relatively high passage rates (Amaral et al., 2012; Bunt et al., 2012; Noonan et al., 2012). Hence, this study is only focused on the potential outcomes resulted from changes in the Milford dam. It has to be noted that this study does not intend to develop a predictive tool of the real alewife populations and hydropower generation in the Penobscot River, but rather to provide an understanding of the potential energy-fish trends and tradeoffs under hypothetical scenarios. This model runs on a daily time step over 200 years.

Figure 4-2 shows an abstracted version of the stock-and-flow diagram of the alewife population model used in this study. The complete version of the SDM is provided in the Appendix C. The alewife model is an age-structured model that mimics the actual life cycle of alewives represented by different age groups. Alewives spend most of their life in the ocean, but spawn in freshwater bodies. Once eggs are hatched, juveniles only live in the freshwater grounds for several months before migrating seaward. In this model, we assume alewives could live up to six years old in the ocean (Messieh, 1977). The adult fish have to reach sexual maturity to participate in the annual spawning runs. We assume the earliest time for an alewife to reach sexual maturity is at age three; however, it is also possible to take longer. The distribution of probabilities in reaching sexual maturity at different ages was obtained from (Gibson and Myers, 2003) and (Barber et al., 2018). Once sexual maturity is reached, the mature fish can participate in multiple spawning runs in the following years until its physical death. Alewife populations at each age group are modelled as stocks. There are primarily two types of stocks beyond the juvenile stock: 1) stocks that keep track of the different age groups in the ocean, and 2) stocks that keep track of number of alewives that enter the spawning runs every year. For the first type of stocks, inflows are surviving alewives returning from the spawning run and surviving alewives from the previous age stock that remain in the ocean. The outflows include alewife loss due to natural mortality in the ocean and advancement to the next age stock. For the second type of stocks, the inflow is the amount of mature alewives from each age group, and the outflows are alewife losses due to natural (i.e., predation) and anthropogenic (i.e., fishing, turbine kill) reasons and advancement to the next age group in ocean. For each spawning run, the number of reproduced eggs is an auxiliary variable which is calculated as a product of three main variables: the number of female spawner, spawner fecundity of each age group, and spawning probability. The population of juveniles is characterized using the Beverton-Holt spawner-recruit curve (Barber et al., 2018; Gibson, 2004). The detailed equations for the model were obtained from (Barber et al., 2018) and (Song et al., 2019) and can be found in Section C3 of the Appendix C.



(Note: FW refers to freshwater)

Figure 4-2. A simplified version of the stock-and-flow diagram showing the key components of the age-structured alewife population model. A full stock-and-flow diagram of the model used in this study can be found in the Appendix C. Variables in boxes are stocks. Arrows with valves are flows in and out of the stocks. Variables without boxes are auxiliary variables. Blue arrows are connectors. The diagram in the green shadow shows spawner upstream migration model (A) and downstream migration model (B).

Figure 4-3 shows an abstracted version of the stock-and-flow diagram of the hydropower generation model (see Section C1 of the Appendix C for the complete model). Reservoir storage behind the Milford dam is a stock variable. Streamflow from upstream that goes into the reservoir is an inflow. Fishway attraction flow, spillway release, and actual turbine release are the three outflows of the reservoir storage. As the dam is a run-of-river dam, we assume the reservoir storage does not change over time. Hence, the inflow always equals to the summed value of the three outflows. The allocation of inflow to the three outflows are based upon rules developed in Song et al. (2019). Hydropower generation at the Milford dam is an auxiliary variable, which is calculated as a product of actual turbine release, net water head, plant overall efficiency, turbine operation period, and two constant variables including water density, 1000 kg/m^3 and gravitational acceleration, 9.8 m/s^2 (Adeva Bustos et al., 2017; Hadjerioua et al., 2012; Power, 2015; Singh and Singal, 2017).

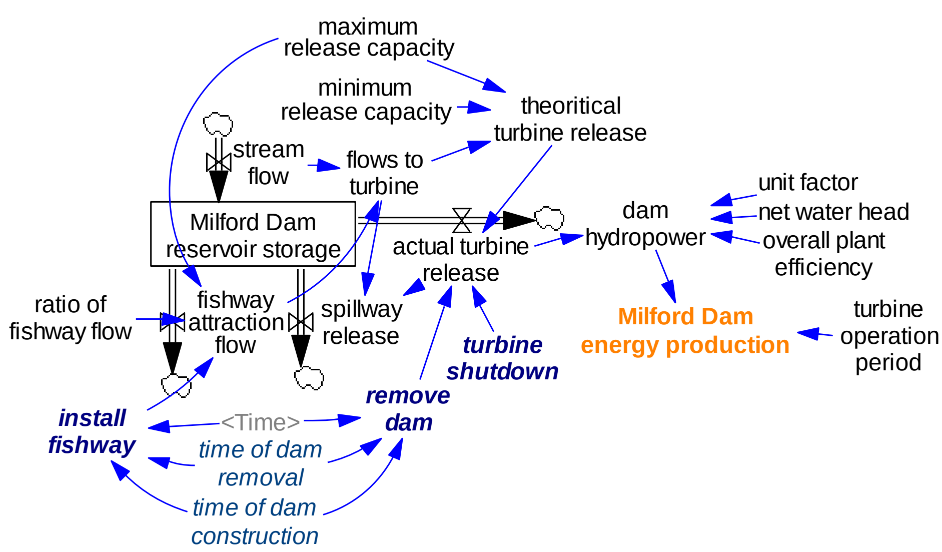


Figure 4-3. Schematic stock-and-flow diagram of energy generation model.

The energy and fish models are connected through changes in the dam's life span. The amount of hydropower generation/loss is roughly linearly related to a dam's life span. The longer the dam is in operation; the more hydropower will be generated. Dam removal will result in a termination in hydropower generation. On the other hand, fish population changes after dam construction or removal are expected to be non-linear. We will investigate the temporal energy-fish tradeoffs under different dam life span scenarios.

4.2.3. Assessed indicators and studied scenarios

The dynamic changes of alewife spawner abundance, loss of fish biomass, and alewife restoration period were analyzed. Alewife spawner abundance in the freshwater habitats is of interest because they are the main source of fishery (Havey, 1961). In addition, spawner abundance and biomass are commonly used indicators to assess the effects of fish-related conservation projects (e.g., stocking program, dam removal) (Pelletier et al., 2008) and the adverse impacts of dam construction (Ziv et al., 2012). In this study, spawner abundance was calculated as the sum of alewife spawners of different age groups that successfully reach freshwater spawning habitats and ready to spawn. Loss of fish biomass was the time summed spawner biomass loss compared to the pre-damming level during dam's life span and the alewife restoration period. Spawner biomass loss was quantified in terms of weight, which is estimated as a product of the loss of spawner populations and their body weights. The average weights of age- 3, 4, 5, and 6 alewife spawners are 144, 186, 209, and 244 g, respectively, according to previously reported alewife trap data in Brunswick, Canada (Barber et al., 2018; Fisheries and Oceans Canada et al., 1981-2016). Alewife restoration period is defined as the period between a specific dam management action (e.g., dam removal) and the full restoration of fish population to the pre-damming level. In this study, the time of full restoration was assumed to be the time point when alewife spawner population reaches 99.5% of the pre-damming level.

Changes of the aforementioned three indicators were investigated under scenarios with different dam life span, upstream passage rate, and size of the downstream accessible habitat area. Dam life span is defined as the total time period that a dam exists in the river channel. We investigated 9 scenarios in dam life span, ranging from 1 to 30 years. The maximum 30-year dam life span is selected considering that the licenses issued by the Federal Energy Regulatory Commission (FERC) for operating non-federal owned hydroelectric dams are usually valid for 30 years (Madani, 2011). During this period, big changes in dam management (e.g., removal, fishway installation) are uncommon. Upstream passage rate is the percentage of fish individuals that are attracted to, enter, and successfully ascend a fishway (Silva et al., 2018). Seven scenarios in upstream passage rate ranging from 0 to 100% were investigated. The range was selected based upon the previously reported effectiveness of fishways (Bunt et al., 2012; Noonan et al., 2012). Size of the downstream habitat area is the accessible spawning habitat areas (in km^2) located downstream of the Milford dam. The current size of the downstream habitat area and the total habitat area in the Penobscot River Basin are 42 and 330 km^2, respectively, according to data collected by Maine Stream Habitat Viewer (MaineDMR, 2017). Ten different sizes of the downstream habitat areas ranging from 0 to 330 km^2 were examined in this study.

4.2.4. Sensitivity analysis

A Monte Carlo simulation (also known as multivariate sensitivity simulation) was performed to understand the influence of parameter uncertainties on alewife spawner abundance (Cheng et al., 2018; Sterman, 1984; Ventana, 2002). All constant variables within the base model were varied by -20% to 20% of their original values as provided in Table 4-1. The Monte Carlo simulation was repeated for 200 times.

Table 4-1. Tested variables in the Monte Carlo simulation

Tested variable	Original value	Test range

Spawning mortality	0.45	[0.36, 0.54]
Fishing mortality	0.4	[0.32, 0.48]
Ocean mortality	0.648	[0.518, 0.778]
Turbine mortality	0.15	[0.12, 0.18]
Fecundity slope	872	[697, 1046]
Fecundity intercept	50916	[40732, 61099]
Alpha	0.0015	[0.0012, 0.0018]
Recruits per HA (age-0 fish/km^2)	811246	[648997, 973495]
Age-3 mature probability	0.35	[0.28, 0.42]
Sex ratio	0.5	[0.4, 0.6]
Total habitat area (km^2)	330	[264, 396]

4.3. Results and Discussions

4.3.1. Energy-fish nexus under different dam life spans

Figure 4-4 (a) depicts the temporal changes of alewife spawner abundance with different dam life spans. For this investigation, we keep the size of downstream habitat area to be 42 km^2 and assume the upstream passage rate at the Milford dam is zero given that there is no fishway installed at this site for at least 60 years (Maynard et al., 2017; Song et al., 2019). According to the results, the undammed Penobscot River could support around 8.45 million of alewife spawners. Construction of the Milford Dam in Year 0 could cause a dramatic decrease in spawner population. This sharp population decline happens mainly within eight years after dam construction. Thereafter, fish population decline slows down and reaches the lowest point at around the 20th year. The spawner population then stabilizes at around 1.10 million, which is 87% below the pre-damming value. It should be noted that spawner population decline does not happen until three years after dam construction. This delayed effect is attributed to the amount of time needed (i.e., 3 years) for the reproduced offspring after dam construction to mature and replace the older generations in the spawning runs. Dam construction blocks alewife spawners' passage to the upstream areas and hence, limits spawning activities to the downstream habitat areas. This limits the amount of offspring being produced due to a higher competition for food and other resources within a smaller size of spawning area. This delayed effect on fish population has been reported and discussed in previous field and modeling studies (Beckerman et al., 2002; Einum and Fleming, 2000; Ford, 2000; Mousseau and Fox, 1998).

Dam life span determines the number of initial fishery populations at the time of dam removal (Figure 4-4 (a)), as well as the time needed to restore alewife population to its pre-damming level (Figure 4-4 (b)). According to the results, dam life span has a significant influence on alewife restoration period within the first 5 years of dam construction. Alewives need 18 years to recover even if they only experience a one-year blockage to the upstream critical habitats. Within 5 years of dam blockage, alewife restoration period has a linear relationship with the duration of blockage, with a 1-year increase in blockage resulting in a 2-year increase in alewife restoration time. If the blockage duration is longer than five years, alewife restoration period will gradually approach and stabilize at 28 years, which is the maximum alewife restoration period needed under the assumed condition. Our results show that the number of initial fish population at the time of dam removal has a vital influence on the alewife restoration time; however, it is only true when the duration of blockage is 5 years or less. This is an extremely short period of time given that most of the dams in the US have a nominal 50 years of designed life span (Ho et al., 2017). Once the threshold (e.g., 5 years for alewife) is passed, the restoration time is no longer sensitive to dam life span. On the other hand, the extensive harm that can be caused by even short periods of passage blockages needs to be recognized and addressed in future river restoration projects.

The energy-fish nexus was analyzed under different dam life span scenarios as shown in Figure 4-4 (c). In our model, the amount of hydropower generation is linearly related to dam life span. However, hydropower generation and loss of fish biomass present a two-segment linear relationship. If dam life span is less than 5 years, generating 1 GWh energy could cause around 0.04 million kg loss of fish biomass, otherwise, the loss of fish biomass is reduced to 0.02

million kg. For a 30-year life span, the Milford dam could provide around 1900 GWh energy with a loss of 45 million kg of fish biomass. The fish biomass loss rate of 0.02 million kg/GWh also applies if the dam life span is longer than 30 years. This is because alewife population stabilizes 20 years after the dam construction, and the annual loss of fish biomass thereafter keeps constant no matter how long the dam life span is. Hence, there are always tradeoffs between hydropower generation and fish biomass losses regardless of dam life span, while the tradeoff is more prominent within the immediate years following dam construction.

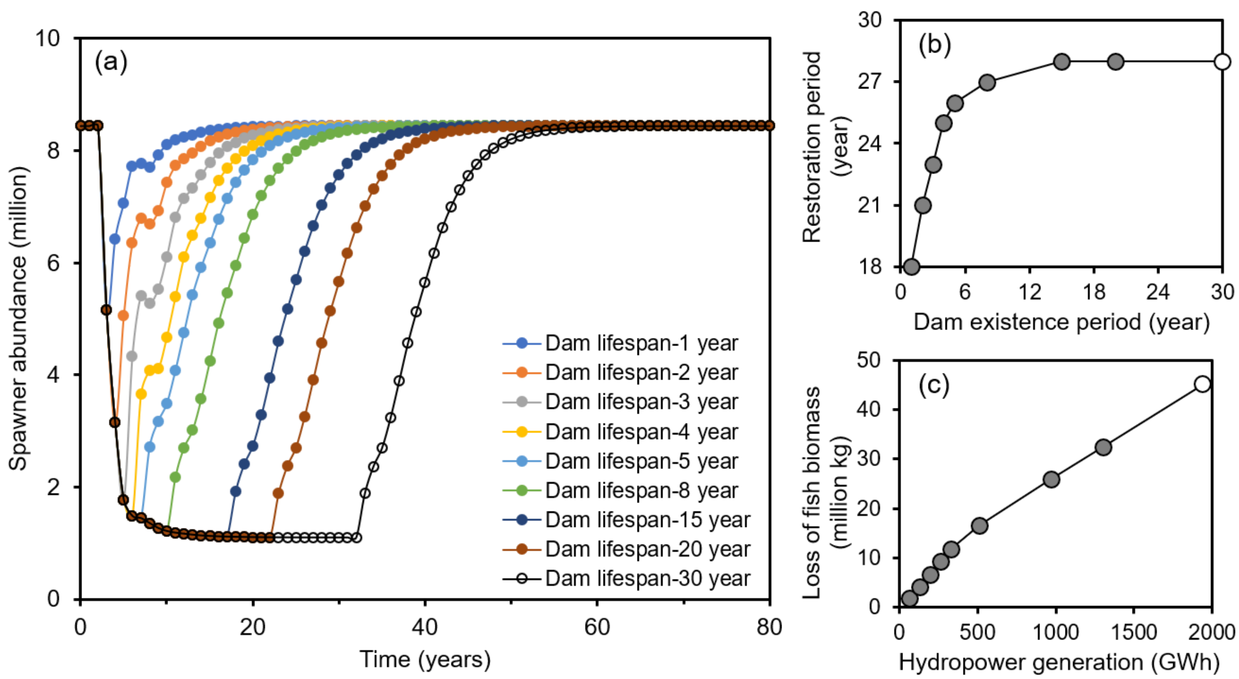


Figure 4-4. Temporal changes of alewife spawner abundance with dam life span (a), relationship between dam life span and fish restoration period (b), and the energy-fish nexus (c).

4.3.2. Energy-fish nexus under different dam upstream passage rates

Dam upstream passage rate determines the number of alewife spawners that can reach the upstream critical habitats. In this section, we keep dam life span and size of downstream habitat area to be 30 years and 42 km^2, respectively, and explore the impacts of different upstream passage rates on temporal changes of alewife spawner abundance (Figure 4-5 (a)). With the improvement of dam upstream passage rate from 0 to 100%, alewife spawner population increases correspondingly from around 1.1 million to around 8.45 million. Conversely, fish restoration period decreases from 28 years to zero year (Figure 4-5 (b)). This is consistent with our previous finding that the more initial fishery population at the time of conducting dam removal, the shorter the restoration period. The relationship between upstream passage rate and alewife restoration period is in a convex shape, which turns at the point with an 80% upstream passage rate. When the dam upstream passage rate is less than 80%, alewife restoration period decreases by around 2.1 years with a 10% increase in the upstream passage rate. When the upstream passage rate is larger than 80%, alewife restoration period decreases by around 5.5 years with a 10% increase in upstream passage rate. Thus, the negative impacts of damming on diadromous fish species could be significantly reduced through installing effective fishways (i.e., >80% upstream passage rate). In practice, however, such a high passage rate is rare. The five commonly used fishways, including pool-and-weir, vertical slot, natural, Denil, and fish lock/elevator, have reported an average upstream passage rate of 61.7% for salmonids, and only 21.1% for non-salmonids (Noonan et al., 2012).

The loss of fish biomass decreases with the increase of dam upstream passage rate in a concave shape which turns at the point of around 60% of upstream passage rate (Figure 4-5 (c)). When upstream passage rate is less than 60%, fish biomass loss is more sensitive to changes in the upstream passage rate. A 10% increase in upstream passage rate at the Milford dam could result in around 6.3 million kg decrease in the loss of fish biomass. This is because a small increase in upstream passage rate under this condition can significantly increase initial fish population at the time of dam removal as shown in Figure 4-5 (a). When upstream passage rate is larger than 60%, a 10% increase in upstream passage rate could only lead to around 1.9 million kg decrease in the loss of fish biomass. From an energy-fish perspective, increasing upstream passage rate is an effective means of balancing the energy-fish tradeoff as it significantly increases fish biomass with minimal loss of energy.

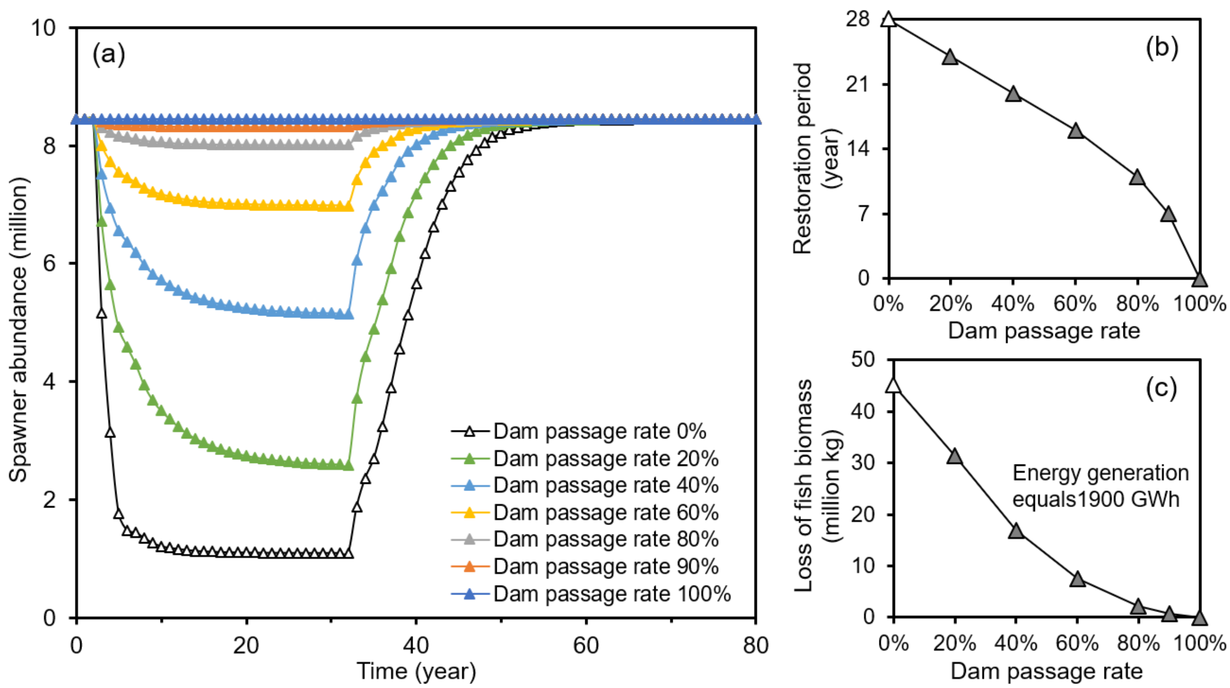


Figure 4-5. Temporal changes of alewife spawner abundance with upstream passage rate (a), relationship between dam passage rate and fish restoration period (b), relationship between dam passage rate and loss of fish biomass (c).

4.3.3. Energy-fish nexus under different sizes of downstream habitat areas

We further examined the response of alewife spawner populations to the sizes of downstream habitat area. Here, dam life span and upstream passage rate are kept constant at 30 years and 0%, respectively. Since the Milford dam is assumed to be totally impassable for alewife, the size of downstream habitat area represents the only accessible habitat areas to alewife spawners. The temporal changes of spawner abundance under different sizes of downstream habitat area are presented in Figure 4-6 (a). When increasing the size of downstream habitat area from 0 to 330 km^2, the stabilized spawner population increases from 0 to 8.45 million. A smaller size of downstream habitat area leads to a lower spawner abundance (Figure 4-6 (a)), and a longer fish restoration period (Figure 4-6 (b)). There is a “S” shaped relationship between the size of downstream habitat area and the restoration period which turns at the sizes of around 16 and 295 km^2 as shown in Figure 4-6 (b). For instance, when the downstream habitat area is 16 km^2 or less, the impaired alewife population needs more than 32 years to restore to its pre-damming population level after 30 years of dam existence. Under the extreme condition where downstream habitat area equals to zero, alewife is likely to extinct in this area after eight years of dam construction. This shows that recovering the threatened or endangered fish species is usually a slow process which would consequently require more efforts and money. Meanwhile, a small increase in the habitat area under this condition can dramatically decrease the length of restoration period. When the downstream habitat area changes between 16 and 295 km^2, a 40.5-km^2 increase in habitat area could steadily decrease fish restoration period at a rate of 2.4 years. When the downstream habitat area further increases to larger than 295 km^2, the restoration period once again becomes sensitive to changes in downstream habitat area. A 40.5-km^2 increase in habitat area can result in an 18.4-year decrease in restoration period.

The loss of fish biomass decreases with the increase of downstream habitat area. The rate of change turns at the point of around 16 km^2 of downstream habitat (Figure 4-6 (c)). The maximum loss of fish biomass over a 30-year dam life cycle is 123 million kg when there is no downstream habitat area available for alewife. This value decreases significantly to around 53 million kg if increasing downstream habitat area to 16 km^2. When the size of downstream habitat area is more than 16 km^2, the loss of fish biomass decreases linearly at a rate of 6.9 million kg with the increase of every 40.5-km^2 downstream habitat area. In order to avoid significant loss of fish biomass, it is important to not build or to remove dams at sites where extremely small size of downstream habitat area is available.

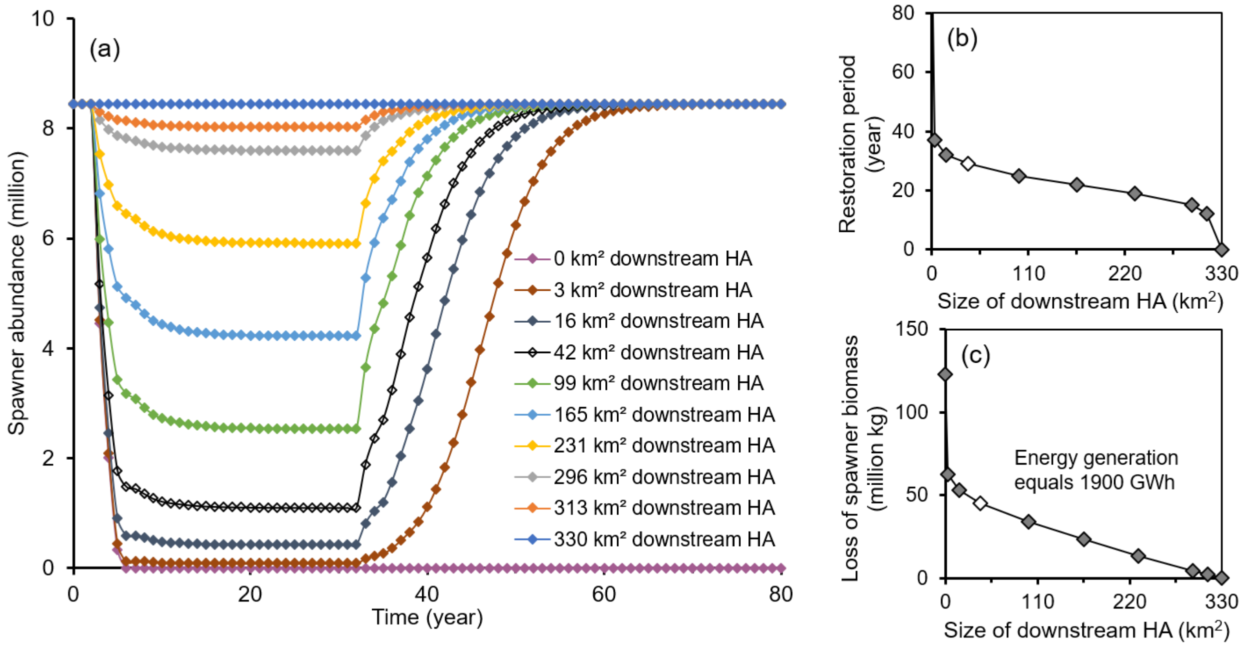


Figure 4-6. Time-series changes of alewife spawner abundance with size of downstream habitat area (HA) (a), relationship between size of downstream habitat area and fish restoration period (b), relationship between size of downstream habitat areas and loss of fish biomass (c).

4.3.4. Sensitivity analysis

The 50%, 75%, 95%, and 100% likelihood of the alewife spawner abundance in response to changes of the tested variables are shown in Figure 4-7. The results show that alewife abundance in the studied river basin stables at a range of 0.9-24.1 million with a 100% confidence, and a range of 5.9-11.8 million with a 50% confidence. With a 30-year blockage of fish passage, we will have 100% confidence that the restoration period will range from 18 to over 90 years, and 50% confidence that the restoration period will range from 26 to 40 years with the changes of the tested model variables. The loss of fish biomass will range from 9.9-118 million kg with a 100% confidence, and 34-64 million kg with a 50% confidence. Our analysis shows that uncertainties in the values of the model variables are likely to result in alewife populations that are susceptible to collapse or an extremely long restoration period. This phenomenon could be explained by the low survival rate at low population levels (Quinn and Collie, 2005). In such a case, fish restoration activities (e.g., lower fishing mortality, fish stocking program) may need to be executed quickly to shorten the restoration period. The results also show that alewife populations reach equilibrium under all scenarios. This is an outcome of the necessary biological process of density dependence, usually in early life history (Quinn and Collie, 2005; Quinn and Deriso, 1999).

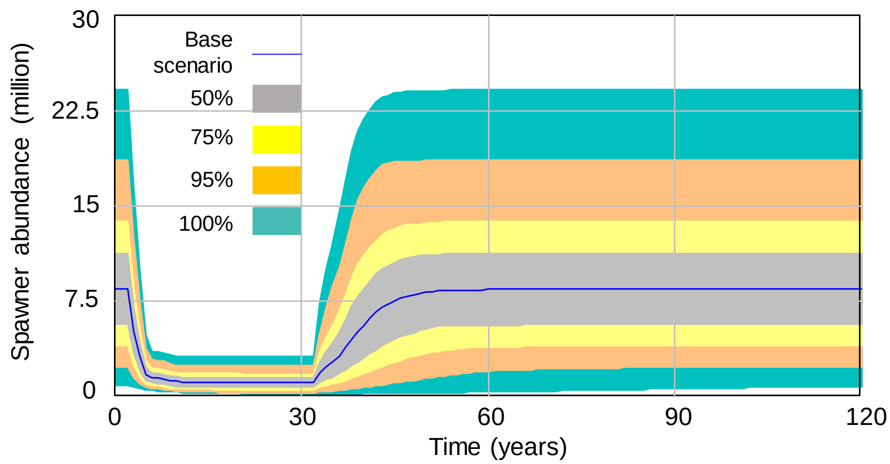


Figure 4-7. Monte Carlo simulation of the age-structured fish population model

4.4. Conclusions

In recent years, a lot of efforts have been made to minimize dams' negative impacts and restore impaired fish populations through fishway installation (Unami et al., 2012), dam removal (Burroughs et al., 2010), and stocking programs (Moring et al., 1995). Our study provides a unique temporal perspective to the hydropower and fish population tradeoffs related to dam life span and initial fishery conditions. Diadromous fish populations are found to be highly sensitive to even a short blockage period. In our modeled river basin, alewives need 18 years to recover to pre-damming level even if they only experience a one-year blockage. Meanwhile, the most dramatic fish population decline happens within five years of dam construction. These findings suggest that dam-related improvement/restoration projects need to be carried out simultaneously with or immediately following dam construction to eliminate dams' impacts on diadromous fish species. From the perspective of energy-food nexus, we found a two-segment linear relationship between hydropower generation and loss of fish biomass under changes in dam life span. When the dam life span is less than five years, generating 1 GWh energy can cause around 0.04 million kg loss of fish biomass; otherwise, the loss of fish biomass is 0.02 million kg. While building hydroelectric dams almost always lead to a fish biomass loss, the effect can be minimized through means such as increasing dams' upstream passage rate, building dams at the sites where large amount of downstream habitat areas is available, or removing dams that significantly block critical upstream habitat areas. Our study shows that a 10% increase in upstream passage rate could reduce fish biomass loss by at least 1.9 million kg in the modelled river basin. Meanwhile, a 40.5-km^2 increase in downstream habitat area can reduce fish biomass loss by more than 6.9 million kg. Both strategies can be achieved with minimal losses of hydropower generation capacities. The Penobscot River Restoration Project is an example case where the energy-fish outcomes were optimized through removing two lower most dams (the Veazie dam and the Great Works dam), improving fish passage performance at the Milford dam, and installing turbine facilities at other existing dams (Opperman et al., 2011).

CHAPTER 5: CRADLE-TO-GRAVE GREENHOUSE GAS EMSSIONS FROM DAMS IN THE UNITED STATES OF AMERICA

5.1. Introduction

The United States of America (USA) has one of the most heavily dammed river systems in the world (Hart et al., 2002; McCully, 2001; Poff and Hart, 2002). More than 90,000 existing "large" dams are documented in the latest National Inventory of Dams (NID) maintained by the Army Corps of Engineers (U.S. Army Corps of Engineers, 2013). This does not include an estimated 2,000,000 or more smaller dams that do not meet the NID criteria for inclusion in the inventory (high or significant hazard classification; 7.6 m in height and exceed 18,500 m^3 in storage; or, 61,700 m^3 storage and exceed 1.8 m in height). The USA also has a long history of building dams. Some of the oldest dams listed in the NID were built in the mid-1600s. The construction of dams continued to grow exponentially thereafter and did not slow down until it peaked in the 1960s (Figure 5-1). In fact, more than one-third of all dams in the NID were built between 1961 and 1980. Dams are constructed for a myriad of primary functions. The primary functions of NID-listed dams are recreation (28.0% of the total number of dams), flood control (17.9%), fishing and fire protection (17.3%), water supply and irrigation (14.7%), power generation (2.3%), erosion control (1.6%), and mine tailings storage (1.3%) (U.S. Army Corps of Engineers, 2013). These primary functions have changed substantially over the years. Most of the dams constructed before the 1900s primarily serve recreational functions currently, although most likely served alternate purposes at the time of their construction. The need for dams for water supply and irrigation became prominent in the late 1800s and the first half of the 1900s, while most dams constructed in the past 50 years are primarily for flood control, fishing, and fire protection. Most of the existing hydroelectric dams (dams capable of generating hydropower) were built between 1800 and 1960; however, hydropower has consistently comprised a small percentage of primary dam functions.

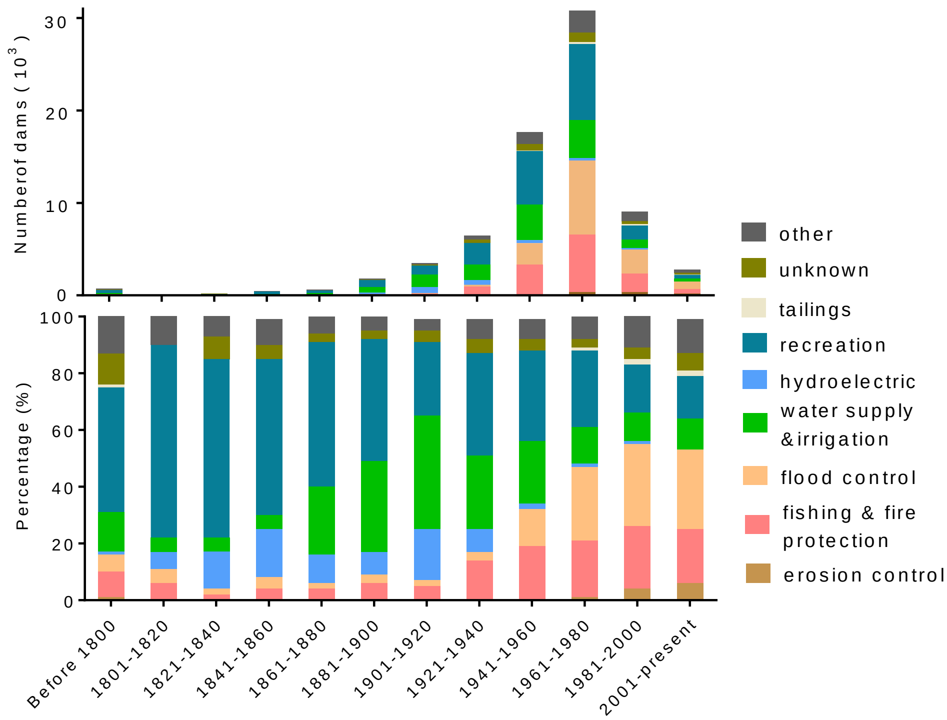


Figure 5-1. The current primary functions of dams constructed in the USA history based on the data obtained from the National Inventory of Dams (U.S. Army Corps of Engineers, 2013)

Although the USA has benefited from the multiple functions provided by dams, their adverse environmental and social impacts and safety risks are increasingly being recognized and debated. For instance, dams have been criticized for altering natural flow regimes, blocking fish passage, affecting sediment transport, and changing watershed characteristics, which collectively contribute to the degradation of water quality, fish population, and biodiversity as well as cascading social and economic problems (e.g., revenue loss in the fishing industry) (Bunn and Arthington, 2002; Gehrke, P.C. et al., 2002; Liermann et al., 2012; Poff et al., 2007; Ziv et al., 2012). Furthermore, some of the older and/or larger dams are often perceived as a public-safety risk under the increasing possibility of natural and man-made threats (Hartford and Baecher, 2004; McClelland and Bowles, 2002). These changes in knowledge have led to a subtle shift in scientific and public attitudes towards dams, and the classification of hydropower as “clean” energy has also been challenged. New dam construction is often accompanied by social opposition, and most importantly, dam removal and upgrades can be contentious, often driven by grassroots movements initiated by local communities (Kosnik, 2010b; O'Connor et al., 2015). Table 5-1 summarizes existing literature on major environmental, social, and economic impacts associated with dams as well as their potential rehabilitation methods.

In the last decade, the method of life cycle assessment (LCA) has increasingly been adopted in assessing the sustainability of products and systems (Guinee et al., 2011; Kloepffer, 2008; Klopffer, 2005). LCA, guided by the ISO 14040 and ISO 14044 standards, is an approach for characterizing the cradle-to-grave or cradle-to-cradle impacts of a product or system, i.e. from raw material acquisition, equipment manufacturing, and use to disposal or reuse (Pryshlakivsky and Searcy, 2013; Varun et al., 2009). Hydroelectric dams, although representing only 2.3% of the total number of dams in the NID, have been the core of most dam-related LCAs (Gallagher, J. et al., 2015b; Varun et al., 2009). This can be partly explained by the significance of hydropower as a type of renewable energy in the USA; hydropower accounts for 6% of the annual USA net electricity generation and 46% of the total renewable energy generation (compared with 35% wind, 2% wood and waste, 1% solar, and 0.4% geothermal) (Cuellar and Herzog, 2015; National Renewable Energy Laboratory, 2013; U.S Energy Information Administration, 2017). Hydropower continues to be developed around the world and holds a critical position in meeting future energy demand, especially in countries where the hydropower potential has not yet been fully exploited (Zarfl et al., 2015). Although new construction of hydroelectric dams has been sluggish since the 1960s in the USA, new programs have been implemented to increase hydropower generation, including (1) development of hydrokinetic energy technologies to extract and convert energy obtained from oceans, rivers, and man-made canals; (2) upgrades of existing hydroelectric dams; and, (3) conversion of existing non-powered dams (dams without hydropower generation capabilities) to hydroelectric dams (Kosnik, 2008; Laws and Epps, 2016; Moreno Vásquez et al., 2016).

Hydropower is traditionally regarded as a low-carbon energy source. Case studies in the USA (Pacca and Horvath, 2002), Canada (Mallia and Lewis, 2013), Japan (Hondo, 2005), Turkey (Atilgan, B. and Azapagic, A., 2016; Atilgan, Burcin and Azapagic, Adisa, 2016), and New Zealand (Rule et al., 2009) compared hydropower with renewable and fossil fuel sources, and found that greenhouse gas (GHG) emissions from the life cycle of hydropower can be as much as 79%, 62%, 88%, and 99% lower than solar photovoltaic (PV), wind, geothermal, and coal, respectively. On the other hand, some studies have suggested that hydropower production could potentially release more GHG emissions than fossil fuel energy from a life cycle perspective, especially considering the large amount of methane emitted from flooded biomass (Deemer et al., 2016; Fearnside, 2016; Hertwich, 2013). Steinhurst et al. (2012) (Steinhurst et al., 2012) estimated that tropical reservoir-based dams could emit 1,300-3,000 g CO_2 eq./kWh, compared to 400-500, 790-900, and 900-1,200 g CO_2 eq./kWh for thermoelectric plants using natural gas, oil, and coal, respectively. Similarly, Fearnside (2015) (Fearnside, 2015a) compared the hydropower generated from the Petit Saut Dam (French Guiana) with electricity generated from combined-cycle natural gas, and found that the GHG emissions from the dam are 19 times higher than the natural-gas-based electricity. The contradictory conclusions of dam GHG emissions reflect our limited understanding of the overall sustainability of hydroelectric dams and the associated implications on the optimal design and operation of these dams. Furthermore, non-powered dams have been largely neglected in previous LCAs despite the large number of such dams.

In this study, a critical review was conducted based on 31 LCA case studies (16 peer-reviewed journal papers) about GHG emissions from hydroelectric dams, 4 additional river in-stream hydropower LCA case studies (2 peer-reviewed journal papers), and more than 20 peer-reviewed journal papers (non-LCA studies) about reservoir GHG emissions. The goal of this study is to understand the significance of life cycle GHG emissions associated with different types of dams, analyze the 'hot-spots' of dam GHG emissions, and identify potential approaches to reduce dam GHG emissions at construction (Section 4), operation and maintenance (Section 5), and demolition (Section 6) stages. In addition, the importance of GHG emissions from reservoirs was analyzed (Section 7). Finally, the life cycle GHG emissions from dams were synthesized and a comparison of hydropower with fossil fuel and other types of renewable energy was performed (Section 8).

Table 5-1. Potential environmental and socioeconomic impacts of dams and prospective amelioration approaches

Potential impacts	Response	Potential rehabilitation tools	Impact assessment methods
Environmental impacts			
Alteration of natural flow regime	Dampening of large or seasonal floods, resulting in a negative impact on both habitat and organisms (Loizeau and Dominik, 2000; Lytle and Poff, 2004)	Allow spring floods; reduce daily fluctuations; create periodic high flows; widen river	Field observation and measurements (Freeman et al., 2001); ecological model (Scheurer and Molinari, 2003)
Barriers to longitudinal fish migration	Fishes killed when they pass through turbine or fish ladder; reduction of fish population and biodiversity; economic losses from fishery	Remove dam; add or improve fish ladders; upgrade to low-impact hydropower generation technology	Field observation and measurements (Fette et al., 2007); Bayesian state-space model (Holbrook et al., 2014; Nieland et al., 2015; Ziv et al., 2012)
Barriers for the drift of organisms	Degradation of water quality; reduction of biodiversity; reduction of property or recreation values	Remove dam	
Blockage of sediment transportation	Accelerated siltation processes; reduction of the vertical connection between the river and groundwater; effects on the benthic community and spawning conditions for fish; reduction of biodiversity (Berkman and Rabeni, 1987; Schalchli, 1995); greenhouse gas (GHG) emissions (Maeck et al., 2013; Pacca, 2007)	Remove dam; widen rivers; manually move sediment from reservoir to downstream	Ecological model for fish biodiversity (Fette et al., 2007; Schalchli, 1995); LCA of sediment contribution to GHG emissions (Pacca, 2007); life-cycle cost analysis of sediment removal and processing system (Qureshi et al., 2015)
Temperature changes	Temperature stratification in the reservoir (Bednarek, 2001); change of downstream temperature when warm or cool water is released	Remove dam; modify dam structure (e.g., change penstocks to allow withdrawal at different reservoir levels; add weirs downstream	Field observation and measurements (Long et al., 1997)
Inundation of terrestrial habitat	GHG emissions from the degradation of inundated biomass; change of local land use patterns; loss of habitat of original inhabitants	Remove dam	Field measurements and empirical models; life-cycle assessment (Pacca and Horvath, 2002)
Socioeconomic impacts			
Involuntary resettlement for some local communities	Economic and cultural shocks and losses of resettling community; poverty and inequity problems	Avoid or minimize involuntary resettlement; improve livelihood of resettling community; encourage public participation and consensus; provide group support (Trussart et al., 2002)	
Waterborne disease from water impoundment schemes	Fatality; economic losses; common in tropical and subtropical regions	Implement prevention strategies and appropriate disease diagnosis; finance medical care (Koch, 2002)	
Reduction of fish population and biodiversity	Reduction of a protein source in the diet; economic losses from fishery; reduction of property or recreation values	Remove dam; add or improve fish ladders; upgrade to low-impact hydropower generation technology	Bayesian state-space model (Holbrook et al., 2014; Nieland et al., 2015; Ziv et al., 2012)
High upfront capital cost	High cost for dam construction, engineering, and design causes public or private economic burdens (Okot, 2013)		Life-cycle cost assessment (Aggidis et al., 2010; Gu et al., 2009)

Risk of dam failure	Economic losses; life loss	Remove/upgrade dam; inspection and maintenance	Risk assessment (Botero-Jaramillo et al., 2015; Su and Wen, 2013)

5.2. Goal and Scope of Published Dam LCAs

All of the 31 LCA case studies reviewed in this study are attributional LCAs, which characterize environmentally relevant flows during a dam's life cycle instead of the change of impacts resulting from possible decisions. Furthermore, the 100-year global warming potential (GWP) was adopted by all of these studies to characterize GHG emissions. Therefore, this same time frame for characterizing GWP was also adopted in the current review. A large variation of life cycle GHG emissions ranging from 0.2 to more than 185 g CO_2 eq./kWh has been reported by previous LCAs (Pacca, 2007; Raadal et al., 2011). Potential reasons for such a wide range of GHG emissions may include discordance in the system boundary adopted and the LCA methodology applied, among others.

Various system boundaries have been adopted by the studies reported in this review (Figure 5-2). All of the dam LCAs reviewed in this paper included raw material extraction, equipment manufacturing, and dam construction stages. Most of the LCA papers also included impacts associated with the operation and maintenance of hydroelectric systems, except for Gallagher et al. (2015) (Gallagher, J. et al., 2015a). Three papers further considered the GHG emissions associated with reservoir flooding and the flooded biomass decomposition (Pacca and Horvath, 2002; Zhang et al., 2007; Zhang, S.R. et al., 2015). Four papers included dam removal and/or decommission (Miller et al., 2011; Pang et al., 2015; Pascale et al., 2011; Suwanit and Gheewala, 2011). Only two papers investigated the GHG emissions associated with the entire life cycle of raw material extraction, equipment manufacturing, construction, operation and maintenance, reservoir flooding, and dam demolition (Denholm and Kulcinski, 2004; Pacca, 2007). No study included GHG emissions from turbine and downstream degassing of supersaturated methane in deep water due to the pressure drop when passing through turbines and flowing at the downstream of dams. Neglecting these GHG emission sources could potentially lead to underestimation of dams' environmental impacts and misguide decision-making about dams (DelSontro et al., 2011; Giles, 2006).

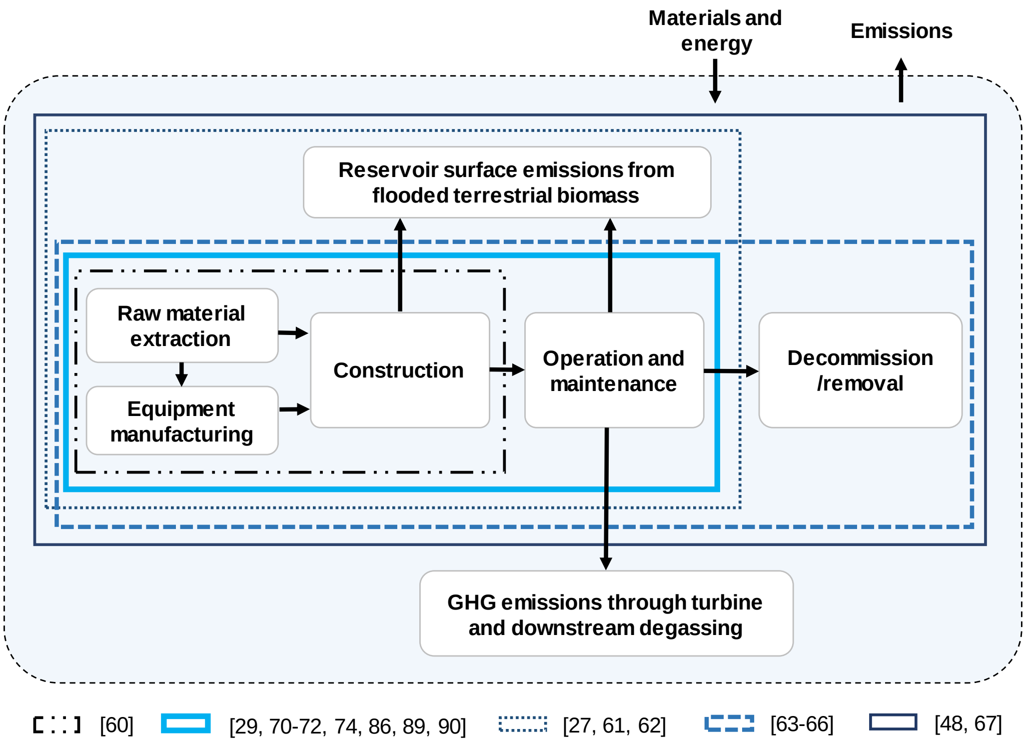


Figure 5-2. System boundaries adopted by previous LCA studies

Three different types of LCA methodologies have been applied in previous dam LCAs, including process-based LCAs (Gallagher, J. et al., 2015a; Pang et al., 2015; Suwanit and Gheewala, 2011), economic input-output (EIO)-LCAs (Varun et al., 2008; Varun et al., 2010, 2012; Zhang et al., 2007), and process-based hybrid LCAs (Liu et al., 2013; Zhang, S. et al., 2015). These methods differ in terms of the amount of upstream processes relevant to a target system that can be included in the analysis. Process-based LCA requires all itemized inputs (e.g., materials, energy) and outputs (emissions) relevant to a dam's life cycle for a complete analysis. As this is difficult to achieve even for the simplest types of products, one often defines a certain boundary of analysis to reduce the amount of data that need to be collected

(Hendrickson et al., 2006; Schenck and White, 2014). EIO-LCA uses EIO tables to characterize the economic interactions among all industries, and hence, no specific boundary decision is required (Hendrickson et al., 2006; Schenck and White, 2014). EIO-LCA often has a broader and more inclusive system boundary than the process-based LCA, but its results are less site-specific due to data aggregation presented in the EIO tables. Process-based hybrid LCA utilizes EIO analysis to supplement process-based LCA for expanding the system boundary. Its system boundary comprehensiveness is often in between the process-based LCA and the EIO-LCA.

5.3. Classification of Hydroelectric Dams and Project

Hydropower projects (HPs) can be classified many different ways: by the quantity of water available (with or without reservoir), available water head (low, medium, or high head), initial installed-electricity-generation capacity (small, large, etc.), or electricity-generation facility type, for instance (Egre and Milewski, 2002; Majumder and Ghosh, 2013). Installed capacity and electricity-generation facility type are the two most common methods used for classification. Most countries set an installed capacity of 10 MW as the demarcation between large and small HPs (Zhang, J. et al., 2015).

Based on electricity-generation facility type, HPs can be divided into four main groups: diversion (run-of-river and canal-based), reservoir-based, pumped storage, and river in-stream HPs (Gaudard and Romerio, 2014). The four types of HPs have different extent and scale of impacts on climate change, different GHG emission “hot-spots” at each of their life cycle stages, as well as different environmental and socioeconomic tradeoffs. For instance, reservoir-based HPs are capable of maximizing energy output through water release control and management and often provide additional services beyond energy generation (e.g. recreation) (Li et al., 2014; Zhao et al., 2014). However, reservoir creation and management is also a significant source of

GHG emissions (Demarty and Bastien, 2011; Huang et al., 2015; Li, S.Y. et al., 2015). Unlike reservoir-based HPs, diversion HPs generally have limited impacts on river flows and do not require creation of large reservoirs. Their life cycle GHG emissions are highly dependent on their structure types, material compositions, and installed capacity (Gallagher, J. et al., 2015a; Gallagher, John et al., 2015). Pumped-storage HPs transfer energy from off-peak to peak hours. They are usually considered energy storage facilities rather than energy generation facilities. In the USA, the total installed capacity of pumped-storage HPs is approaching 21.9 GW, which represents around 97% of the utility-scale electricity storage in the entire nation (Deane et al., 2010). Even though pumped-storage HPs play an important role in electricity storage, limited studies have assessed their environmental impacts, especially considering their unique requirement of two reservoirs for operation. The structure of river in-stream HPs is relatively simple and primarily comprises turbines, power cable, and onshore facilities. There is no need to build dams or weirs, pipelines, or reservoirs for river in-stream HPs. In the USA, river in-stream HPs are mainly installed along the Mississippi River system (Skone, 2012). Among the reviewed LCA studies, eight studied diversion HPs (Gallagher, J. et al., 2015a; Hondo, 2005; Pang et al., 2015; Pascale et al., 2011; Suwanit and Gheewala, 2011; Varun et al., 2008; Varun et al., 2010, 2012), six included reservoir-based HPs (Liu et al., 2013; Pacca, 2007; Pacca and Horvath, 2002; Ribeiro and da Silva, 2010; Zhang et al., 2007; Zhang, S. et al., 2015), two investigated pumped-storage HPs (Denholm and Kulcinski, 2004; Oliveira et al., 2015), and two studied river in-stream HPs (Gallagher, John et al., 2015; Miller et al., 2011). Table 5-2 provides the definition, components, functions, pros and cons, as well as the related LCA studies for the four types of HPs.

Table 5-2. Comparison of the four types of hydropower projects based on electricity-generation facility type

Type of hydropower projects (HPs)	Definition	Components	Primary functions	Pros	Cons	Life cycle studies
Diversion HPs (run-of-river and canal-based HPs)	A facility that channels flowing water from a river through a tunnel or pipeline to power turbines (International Hydropower Association)	Dam/weir, feeder channel, forebay, penstock, powerhouse, electro-mechanical equipment*	Power generation	Limited social and environmental impacts; river flow pattern remains unchanged	Electricity output varies with the river's natural flow	(Gallagher, J. et al., 2015a; Hondo, 2005; Pang et al., 2015; Pascale et al., 2011; Suwanit and Gheewala, 2011; Varun et al., 2008; Varun et al., 2010, 2012)
Reservoir-based HPs	A large system that uses a dam to store water in a reservoir (International Hydropower Association)	Dam, penstock, powerhouse, electro-mechanical equipment*	Recreation; water supply; fire protection; flood control; power generation	Steady power output; deliver multiple services	Social and environmental impacts for local community and the whole watershed; alteration of the ecosystem and natural habitats; displacement of local communities, etc.	(Liu et al., 2013; Pacca, 2007; Pacca and Horvath, 2002; Ribeiro and da Silva, 2010; Zhang et al., 2007; Zhang, S. et al., 2015)
Pumped-storage HPs	Projects harness water that is cycled between a lower and upper reservoir by pump (International Hydropower Association)	One or more dams, penstock, electro-mechanical equipment*, pump, powerhouse	Water supply; fire protection; flood control; power generation	Load following, peaking power, and standby reserve (Denholm and Kulcinski, 2004)	Energy consumption; low efficiency	(Denholm and Kulcinski, 2004; Oliveira et al., 2015)
River in-stream HPs	Projects that generate electricity from the flow of inland waterways (International Hydropower Association)	Turbines, power cables, onshore facility	Power generation	Limited social and environmental impacts	Electricity output varies with the river's natural flow	(Gallagher, John et al., 2015; Miller et al., 2011)

*Electro-mechanical equipment includes turbine, generator, switchgear, control and protection equipment, electrical and mechanical auxiliaries, transformer and switch-yard equipment.

5.4. The Construction Stage of Dams

The construction stage is defined as the raw material extraction, equipment manufacturing, transportation, and actual building processes of dams (each will be discussed further in Sections 5.4.1-5.4.3). It has been estimated that around 2.3 to 37.9 g CO_2 eq./kWh are emitted from the construction stage based on GHG emissions from 27 dams worldwide (Pacca, 2007; Varun et al., 2008; Zhang et al., 2007). Table D1 of the Appendix D provides the GHG emissions associated with each individual contributor to the construction stage. Generally, the construction stage contributes more than 70% and around 50% of dams' total construction and operation emissions (reservoir-related and demolition emissions excluded) based on results from process-based LCAs (Dones et al., 2003; Gallagher, J. et al., 2015a; Pang et al., 2015; Suwanit and Gheewala, 2011) and EIO-LCAs, respectively. The assumptions of dam life span also influence the emission results from this stage. For instance, Hondo (2005) (Hondo, 2005) found an 83% decrease in life cycle GHG emissions (from 30 to 5 g CO_2 eq./kWh) when the lifetime of a 10 MW run-of-river dam is changed from 10 to 100 years. The life span reported by the previous dam LCAs ranges from 20-150 years (Table D1 of the Appendix D). Given that the life span of dams could vary based upon factors such as dam functions, structures, and geographical locations, we adopted the originally reported life-span values in this review. The significant consumption of materials, equipment, energy, and labor makes the construction stage an important GHG emission source for dams.

5.4.1. Raw material extraction and equipment manufacturing

A typical dam structure includes the dam core, pipelines, powerhouse, turbine, and generator. Based on structure design, dams can be divided into four groups: embankment, arch, gravity, and buttress dams. The simplified sectional view of the four types of dams is shown in

Figure 5-3. Embankment dams come in two types: earth dams and rock-filled dams, constructed mainly by earth and rock, respectively. The cross section of an embankment dam has a hill-like shape (Chen et al., 2014; Zhang et al., 2016). Gravity dams are mainly fabricated from concrete and stone masonry, with a triangular cross section (Zhang et al., 2013). The weight of the dam is used to hold back large volumes of water. Buttress dams are made from concrete and masonry. They have a watertight upstream side supported by a series of triangular-shaped walls (buttresses) on the downstream side (Kougias et al., 2016). Arch dams are curved in the shape of an arch, with its convexity towards the upstream side. The cross section of an arch dam is comparatively thinner than a similar-scale gravity dam (Lin et al., 2015). In the USA, embankment dams are predominant and account for about 86% of all dams in the NID database, followed by gravity dams (3.4%).

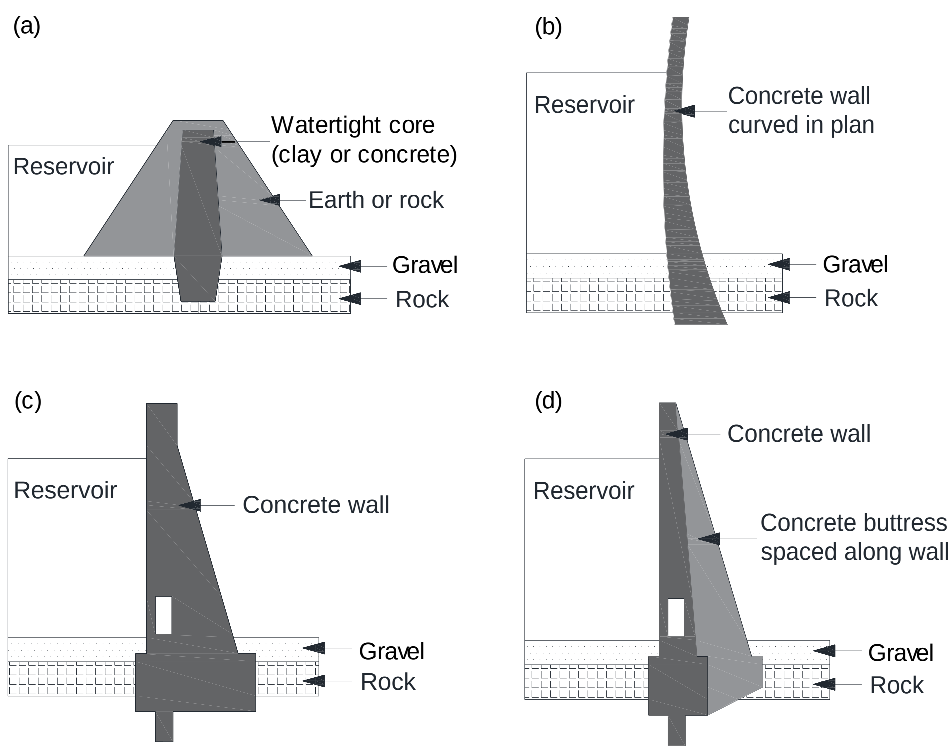


Figure 5-3. The sectional view of four types of dams: (a) embankment dam, (b) arch dam, (c) gravity dam, (d) buttress dam (adapted from the British Dam Society (The British Dam Society, 2012))

Dam structures influence both the quantities and the types of materials needed to build the dam and the associated emissions. For example, buttress dams generally require smaller amounts of construction materials compared to similar-scale gravity dams because of the clear spaces between buttresses (Novak et al., 2007). Embankment dams usually require more construction materials than similar-scale arch, gravity, and buttress dams because of their larger structural volumes (Novak et al., 2007). However, they may have lower GHG emissions because sand and rock used for embankment dams have significantly lower GHG emission factors than those of cement and concrete used for constructing gravity and buttress dams (Liu et al., 2013). Zhang et al. (2015) estimated the life cycle GHG emissions of an earth-rockfill embankment dam and a similar-scale concrete gravity dam, and found that the embankment dam has around 46% fewer raw-material GHG emissions compared to the gravity dam (Zhang, S. et al., 2015). Table 5-3 provides the typical quantities of common materials used to build HPs, their associated GHG emission factors, and the average typical GHG emissions of each material.

Table 5-3. GHG emission factors and typical quantities for different materials

Materials	Application	Typical quantity (kg/MWh)	Emission factor (kg CO_2 eq./kg of material)	Average GHG emissions (kg CO_2 eq./MWh)
Steel	Dam framework; Penstock	0.5 (Pang et al., 2015; Ribeiro and da Silva, 2010; Suwanit and Gheewala, 2011)	2.2 (Zhang, S. et al., 2015)	1.1
Cement	Dam body (arch, gravity, buttress) or dam core (embankment); Penstock	8.3 (Pang et al., 2015; Ribeiro and da Silva, 2010; Suwanit and Gheewala, 2011)	0.9 (International Panel on Climate Change (IPCC), 2006)	7.1
Polyvinyl chloride	Penstock	2.9 (Pascale et al., 2011)	1.8 (National Renewable Energy Laboratory, 2012)	5.1
Sand	Dam body (embankment)	11.0 (Ribeiro and da Silva, 2010; Suwanit and Gheewala, 2011)	0.002 (European Commission Joint Research Centre (ELCD), 2009)	0.02
Gravel & rock	Dam foundation	16.6 (Ribeiro and da Silva, 2010; Suwanit and Gheewala, 2011)	0.002 (European Commission Joint Research Centre (ELCD), 2009)	0.03

Note: Average GHG emissions (kg CO_2 eq./MWh) = Typical quantity (kg/MWh) × Emission factor (kg CO_2 eq./kg of material)

The aforementioned studies have mainly been focused on hydroelectric dams, while the raw-material GHG emissions associated with the large number of non-powered dams remain unknown. As a preliminary attempt to address this knowledge gap, a comparison of the total hydroelectric versus non-powered dams was carried out using dams located in the USA as a case study. In Figure 5-4, the product of dam height and length (perpendicular to river flow direction) was used as a surrogate of dam size and construction material quantities. We calculated the product of dam height and length for each dam in the NID, and summed the products for each of the four dam structure types (Figure 5-4). Within each structure type, we further divided the results into two groups: hydroelectric and non-powered dams. This comparison relies on two critical assumptions. First, the material composition and design variations within each dam structure type are neglected. Second, the influence of dam width variations (parallel to river flow direction) on the quantities of construction materials needed is assumed to be the same for all dams. The results show that there are relatively few arch and buttress dams in the USA, and they have relatively low height × length values for non-powered and hydroelectric dams, indicating their limited overall raw material usages and associated emissions. The total height × length value of the embankment dams is up to 240 times greater than the other three structure types combined, indicating a popularity of embankment dams in the country. Furthermore, the non-powered embankment dams have a significantly higher total dam height × length value than that of the hydroelectric dams (13 times larger), indicating the importance of non-powered dams in material consumption and contributions to raw-material GHG emissions. The results also indicate that hydroelectric dams generally have a larger size than the non-powered dams.

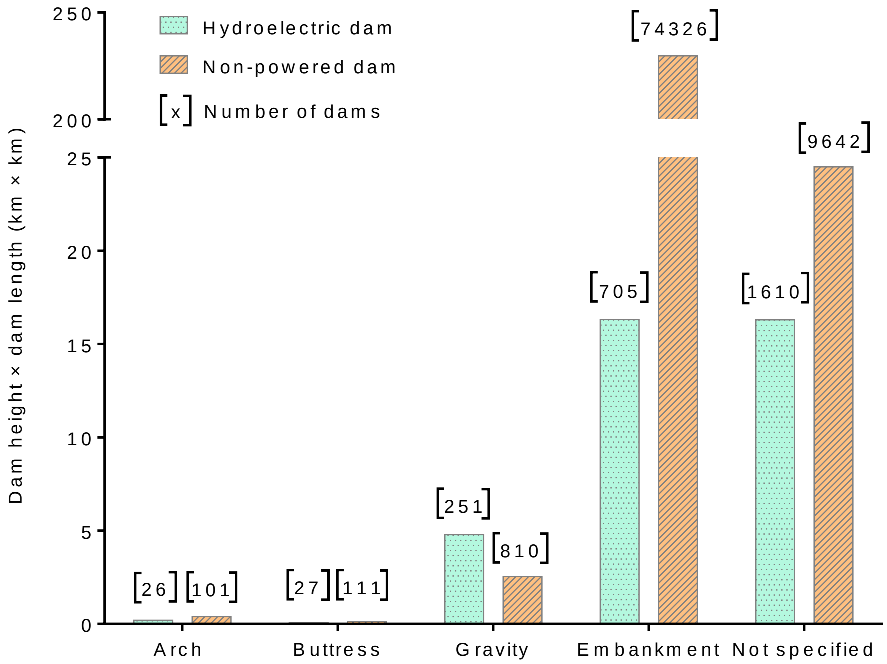


Figure 5-4. The summed value of dam height times dam length (a surrogate value of the total construction-material requirement) for each type of dam in the USA based on NID database

Linking dam structures to hydropower-generation facility types, reservoir-based HPs are usually large embankment and gravity dams. Construction of these dams requires a large amount of materials, which dominates their total construction GHG emissions (including raw material extraction, equipment manufacturing, transportation, and actual construction) (Zhang et al., 2007; Zhang, S. et al., 2015). On the other hand, unlike the large reservoir-based HPs, diversion HPs are usually small and mainly function as a river-diversion channel to penstocks for electricity generation. Hence, pipeline manufacturing is another major contributor to the total construction GHG emissions of diversion dams given that they are usually made of carbon-intensive steel or polyvinyl chloride (PVC) materials (Gallagher, J. et al., 2015a; Hondo, 2005; Pang et al., 2015; Pascale et al., 2011; Suwanit and Gheewala, 2011). Gallagher et al. (2015) calculated the environmental impacts of three small-scale run-of-river HPs in the UK, and found that polyethylene pipework accounted for around 53-60% of the total construction GHG emissions, followed by turbine and generator (19-23%), and powerhouse (13-17%) (Gallagher, J. et al., 2015a). Other construction materials, such as earth and concrete, only present a very small portion of the total construction GHG emissions. Similarly, a case study of a 10 MW run-of-river HP in Japan found that around 39.8% of the construction and operation GHGs come from the penstock (Hondo, 2005).

The importance of material type and quantity in dam construction suggests that reduction of material consumption, design optimization, and utilization of recycled or green materials could be potentially viable ways to improve dams' sustainability (Gallagher, J. et al., 2015a; Pang et al., 2015). Gallagher et al. (2015) examined a number of eco-design measures for the installation of small hydropower plants ranging from 50 to 650 kW, including replacement of concrete-block cavity walls with wooden-frame super-structures for the powerhouse, replacing a fraction of the aggregate or cement with increased recycled content, and using biofuels for onsite machinery and transportation. The results showed that these eco-design measures led to a cumulative reduction of 2.1-10.4% of the total construction GHG emissions (Gallagher, J. et al., 2015c).

5.4.2. Transportation

GHG emissions at the transportation stage are mainly from the consumption of fuel by truck, train, ship, or plane (Horvath, 2006; Zhang, S. et al., 2015). The total weight of transported goods, travel distances, and the types of transportation mode used are the major factors influencing GHG emissions at the transportation stage (Zhang, S. et al., 2015). A wide variation from 0.06 to 5.6 g CO_2 eq./kWh was estimated by previous LCA case studies. Of all LCA's reviewed in this study, only four papers reported the transportation GHG emissions separately in their analysis (Gallagher, J. et al., 2015a; Pang et al., 2015; Suwanit and Gheewala, 2011; Zhang, S. et al., 2015), while other studies combined the impacts of transportation with raw material extraction or actual construction. Of these studies that reported transportation GHG emissions separately, six case studies suggested that transportation only has a marginal impact of less than 3% of the construction GHG emissions (Gallagher, J. et al., 2015a; Pang et al., 2015; Zhang, S.R. et al., 2015). However, a study of five run-of-river HPs located in Thailand found that around 32% of life cycle GHG emissions are from transportation (Suwanit and Gheewala, 2011). This is mainly because the pressure pipelines and electro-mechanical equipment have to be imported from overseas through a long distance to the construction site. Collectively, these varied estimates indicate that localization of material and equipment production is essential to reduce transportation-related environmental impacts (Pang et al., 2015). In addition, utilization of alternative and renewable energy sources for transportation could also potentially reduce GHG emissions.

5.4.3. Actual building and construction processes

GHG emissions during the actual dam-building process are usually combined with the impacts of raw material extraction and equipment manufacturing. Among the 31 dam LCA case studies reviewed, only 9 case studies provided the GHG emissions of the actual building process separately, with results ranging from 0.06 to 11 g CO_2 eq./kWh. The construction of HPs is a complicated process, which includes procedures like excavating, dam filling, concrete mixing, drilling, and blasting (Liu et al., 2013; Zhang, S. et al., 2015). The process of reservoir flooding for reservoir-based dams is not included in this section and will be discussed separately in Section 7. GHG emissions during the building and construction process are mainly from diesel fuel and electricity consumption by on-site equipment installation and usage (Zhang, S. et al., 2015). A previous LCA found that GHGs generated by a conventional concrete dam during actual construction are around 50% higher than a similar-scale rockfill dam mainly because the building of conventional concrete dams requires larger amounts of electricity and oil by cable cranes, air compressors, and dump trucks (Liu et al., 2013). Other factors, such as hydrologic conditions, hydraulics, soil and sediment characteristics, HP designs, and construction techniques, will influence the workload and hence the GHG emissions of the building process (Han et al., 2012; Pang et al., 2015; Suwanit and Gheewala, 2011).

5.5. Operation and Maintenance of Dams

GHG emissions during the operation and maintenance (O&M) stage are mainly associated with the O&M of civil structure and electro-mechanical equipment, consumption of thermal back-up power due to variable electricity generation, and reservoir GHG emissions (further discussed in Section 7). Maintenance of civil structure includes activities such as repairing cracks in the dam body, powerhouse and other civil works, as well as replacing pipework and screen filters. Maintenance of electro-mechanical equipment mainly includes replacement of generators and turbines, changing lubricant oils, and replacing seal plates. A wide range from 0.9 to 77 g CO_2 eq./kWh has been reported by previous LCAs. Some of the important causes of such a wide range include adoption of different LCA methodologies and the wide variance of GHG emissions from reservoirs. For instance, an EIO-LCA of a run-of-river dam with an installed capacity of 3,000 kW in India reported a GHG emission of 18.7 g CO_2 eq./kWh at O&M stage (Varun et al., 2008). In comparison, a process-based LCA of a run-of-river dam with an installed capacity of 3,200 kW in China reported a much smaller O&M GHG emission of 0.9 g CO_2 eq./kWh (Pang et al., 2015). Among the LCAs reviewed, EIO-LCA is a commonly used method to assess GHG emissions of the O&M stage due to the unavailability or difficulty in obtaining detailed historical O&M data of the dams.

Additionally, the match between dams' installed capacity and the available hydraulic capacity will also influence the GHG emissions at the O&M stage. The optimal installed capacity was commonly determined by comprehensive evaluations of historical hydrology data and predictions of the future change of water resource before construction. However, uncertainties of future climate and inaccuracies in these predictions may lead to under-installed capacity and longtime over-loaded operations, accelerating equipment exhaustion and failures. On the contrary, if the available water resource is overestimated, more installed capacity than necessary will be constructed, leading to waste of installed capacity or idling (Pang et al., 2015).

5.6. End-of-life of Dams

The end-of-life of dams usually includes the decommissioning of construction components, and recycling valuable metals and equipment. There have been three different ways to deal with the end-of-life stage by previous LCAs. Most previous LCAs simply exclude the demolition stage due to a lack of data. Some argued that most dams remain for preserving the adapted ecosystems and environments, even though they no longer produce hydropower (Gallagher, J. et al., 2015a; Zhang et al., 2007). Neglecting the end-of-life stage could potentially lead to underestimation of dams' GHG emissions, given that dam removal has a large impact on the release of GHGs from accumulated sediments (Pacca, 2007). A few other studies estimated the GHG emissions associated with the removal of major dam components, such as concrete structures, powerhouse structures, pipelines, and electricity machines, and with the recycling of high-value materials, such as steel, stainless steel, and iron (Pang et al., 2015; Suwanit and Gheewala, 2011). GHG emissions were calculated based on the energy consumption of the demolition machines and material transportation to the landfill or recycling sites. End-of-life GHG emissions in this case were estimated to be low enough to be neglected. Only one LCA paper considered the decomposition of organic matter in the sediment after dam removal (Pacca, 2007). This study pointed out that the decomposition of sediments could generate around 35-380 g CO_2 eq./kWh based on data collected from six LHPs located in the USA with an installed capacity ranging from 185 to 2,000 MW, which is around 18-65 times larger than its construction GHG emissions and 3-26 times larger than the O&M GHG emissions (including the reservoir emissions) (Pacca, 2007). Yet, the ripple effects of ecosystem interruptions after the dam removals, such as downstream fish kills, destabilization of stream banks, and fill-in of riffle-pool habitat, were still not included (Pacca, 2007). Furthermore, there remains a lack of data and studies on the GHG emissions associated with large dam removals, as most of the dams that have been removed in the USA are small dams with a height lower than 4 m (Ryan Bellmore et al., 2016).

5.7. Reservoir GHG Emissions

Decomposition of flooded biomass and organic materials generates carbon dioxide and methane in both aerobic and anaerobic conditions after impoundment. Some of these GHGs emit to the atmosphere through diffusion (CO_2 and methane) or ebullition (methane) at the reservoir surface. These diffusive GHG emissions have been included in LCAs such as Pacca and Horvath (2002) (Pacca and Horvath, 2002), Zhang et al. (2007) (Zhang et al., 2007), and Zhang et al. (2015) (Zhang, S. et al., 2015). However, reservoir GHG emissions happen not only at the reservoir surface, but also when water passes through turbines or spillways, and downstream of dams (Hertwich, 2013). Water passing the turbine is drawn from certain depths of the reservoir. The deeper the water is, the higher the pressure and the lower the temperature becomes. In stratified systems where density boundaries limit the mixing of GHGs, the solubility and concentration of GHGs become higher at greater depth in the reservoirs. When the supersaturated water passes through the turbine, the sudden pressure drop could result in direct release of GHGs into the air. Another part of GHGs are gradually released through diffusion or bubbling downstream of the dam after passing through the turbine. Kemenes et al. (2007) measured that around 39 Gg CO_2 eq. were emitted annually through turbine degassing and downstream emissions at the Balbina dam (Brazil), whereas 34 Gg CO_2 eq. were generated annually at the reservoir surface (Kemenes et al., 2007). De Faria et al. (2015) (de Faria et al., 2015) estimated that GHG emissions through turbine and downstream degassing are around three times the GHG emissions from reservoir surface. Reservoir GHG emissions have been widely studied outside of the LCA field (de Faria et al., 2015; Rosa and Schaeffer, 1995). Table D2 of the Appendix D provides the estimated GHG emissions from the previous studies' aforementioned pathways.

Under the IPCC guidelines, it is an option rather than a requirement to include reservoir GHG emissions for dam LCAs because of three main difficulties with measuring and estimating such emissions (Demarty and Bastien, 2011; Fearnside, 2015a). First, methane is usually produced through anaerobic digestion in sediments and rises up as bubbles. It is hard to accurately measure methane ebullition since bubbles happen in bursts rather than a steady flow (Chen et al., 2009; Chen et al., 2011; Kemenes et al., 2007; Li, S.Y. et al., 2015). Second, factors such as the amount and carbon content of flooded biomass and reservoir productivity often influence reservoir GHG emission rates (Deemer et al., 2016). HPs in humid tropical regions typically have higher GHG emission rates because of larger unit biomass quantities, higher average biomass carbon contents, and warmer temperatures accelerating the decomposition process (Varun et al., 2009). Flooded biomass per unit of reservoir area has been shown to vary from 10 kg/m^2 in boreal regions to 50 kg/m^2 in tropical forests, and carbon content varies from 0.3 kg CO_2 eq./m^2 for desert shrubland to 18.8 kg CO_2 eq./m^2 for tropical forests (Gagnon et al., 2002). GHG emissions from tropical reservoirs have been reported to be around 2-13 times higher than temperate reservoirs (Louis et al., 2000), and around 3-26 times higher than boreal reservoirs (Zhang, J. et al., 2015). In addition, older reservoirs tend to have a lower GHG emission than newly created ones because of the depletion of the labile flooded biomass and soil organic carbon over time (Barros et al., 2011; dos Santos et al., 2006; Louis et al., 2000). Hence, site measurements of specific dams are often difficult to generalize or to apply directly to other dams. Third, different emission pathways dominate depending on reservoir depth (Li, S. et al., 2015). In stratified deep waters (>7 m) where anaerobic conditions prevail, decomposition of organic matter might result in a higher ratio of methane production. Thus, the deeper the electricity generation turbines are located in the water, the more methane will be emitted when water passes through the turbine and flows downstream.

Additionally, reservoir emissions associated with the non-powered dams have been largely neglected. Given the large number of reservoir-based, non-powered dams, understanding the relative scale and importance of their GHG emissions is imperative. Accordingly, we provide a comparison of the total reservoir GHG emissions from hydroelectric dams and non-powered dams in the USA and the results are presented in Table 5-4. Reservoir GHG emission rates in different climate zones were directly obtained from previous reservoir studies (Barros et al., 2011; dos Santos et al., 2006; Hertwich, 2013; Li, S. et al., 2015; Louis et al., 2000). Total reservoir surface area in each climate zone was calculated based on NID data (natural lakes excluded). Around 5% of the total dams did not report their functions, and hence they are excluded from this analysis. Table 5-4 indicates that the total reservoir GHG emissions of non-powered dams are as important as those of hydroelectric dams.

Table 5-4. GHG emissions from total reservoir-based hydroelectric and non-powered dams in the USA based on NID data

Climate zone	Reservoir GHG emission rate* (g CO_2 eq./m^2/yr)	Reservoir surface area (km^2)		GHG emission (Tg CO_2 eq./yr)	
		Hydroelectric dam	Non-powered dam	Hydroelectric dam	Non-powered dam
Boreal	873 (Barros et al., 2011; dos Santos et al., 2006; Hertwich, 2013; Li, S. et al., 2015; Louis et al., 2000)	54	30	0.05	0.03
Temperate	557 (Barros et al., 2011; dos Santos et al., 2006; Hertwich, 2013; Li, S. et al., 2015; Louis et al., 2000)	48,374	51,291	26.94	28.57
Tropical	2,733 (Barros et al., 2011; dos Santos et al., 2006; Hertwich, 2013; Li, S. et al., 2015; Louis et al., 2000)	16	22	0.04	0.06
Total		48,444	51,343	27.03	28.66

*Reservoir GHG emission rates adopted are gross reservoir surface GHG emission rates averaged for the three climate zones based upon previously reported values

Net reservoir emission is another way to quantify reservoir GHG emissions. It is defined as the gross reservoir GHG emissions minus baseline GHG emissions before reservoir creation (Unesco, 2010). Baseline GHG emissions before flooding can either be positive (source) or negative (sink) depending on prior land use. For instance, boreal and temperate forests on average absorb 2,100 mg/m^2/d of CO_2 and 1.0 mg/m^2/d of methane (Fan et al., 1998; Savage et al., 1997) and hence have negative baseline GHG emissions. Lakes have a positive baseline GHG emission of 1,180 mg/m^2/d of CO_2 (Raymond et al., 2013; Tranvik et al., 2009) and 46 mg/m^2/d of methane (Bastviken et al., 2011; Tranvik et al., 2009). When the forests are flooded to form lakes, the resulting net reservoir emissions will be 3,280 mg/m^2/d of CO_2 and 47 mg/m^2/d of methane. Pacca and Horvath (2002) reported that the loss of baseline GHG absorption capacity alone could contribute 7-13% to a dam’s life cycle GHG emissions (Pacca and Horvath, 2002). Besides, the creation of dams also alters the carbon cycle in the original river flow by trapping suspended materials behind the dams (Hertwich, 2013; Maeck et al., 2013). Mendonca et al. estimated that carbon burial could potentially outweigh the carbon emissions from the reservoir surface (Mendonca et al., 2012; Mendonca et al., 2014), yet dam removal will release those trapped sediments which may result in GHG emissions (Pacca, 2007). Nevertheless, this net effect of burial and releasing of GHGs from the trapped sediments has not been included in current dam LCAs. Overall, our understanding of dams' impact on the global carbon cycle is still limited and more research is needed in this area for more accurate quantifications.

5.8. Life Cycle GHG Emissions of Dams

The synthesized values of life cycle GHG emissions from different types of dams are shown in Figure 5-5. Additionally, numerical values of GHG emissions from each life cycle stage provided by previous LCAs and reservoir emission studies are presented in Table D3 of the Appendix D. According to Figure 5-5, pumped storage dams have significantly higher O&M emissions than other types of dams. This is mainly due to the large amount of energy needed by pump operation. Demolition GHG emissions could contribute significantly to the boreal and temperate reservoir-based and pumped storage dams. Reservoir GHG emissions have the largest contribution to the tropical reservoir-based and pumped storage dams. However, boreal and temperate reservoir GHG emissions could be underestimated due to a lack of studies linking these emissions to hydropower productions (Deemer et al., 2016). Reservoir-based HPs are generally much more carbon intensive than diversion HPs. Upstream impoundment emissions, turbine degassing, and downstream emissions from diversion dams have rarely been studied and hence are excluded from Figure 5-5. Fearnside (2013) and Fearnside (2015) provided the only impoundment GHG emission estimation of 63 g CO_2 eq./kWh for a tropical run-of-river dam (Fearnside, 2013; Fearnside, 2015b). Given the importance and large variability of reservoir GHG emissions, more attention needs to be paid to reservoir GHG emissions when decisions have to be made for the development of dams, especially in tropical regions, as most of the future expansion of hydropower is likely to happen in these areas.

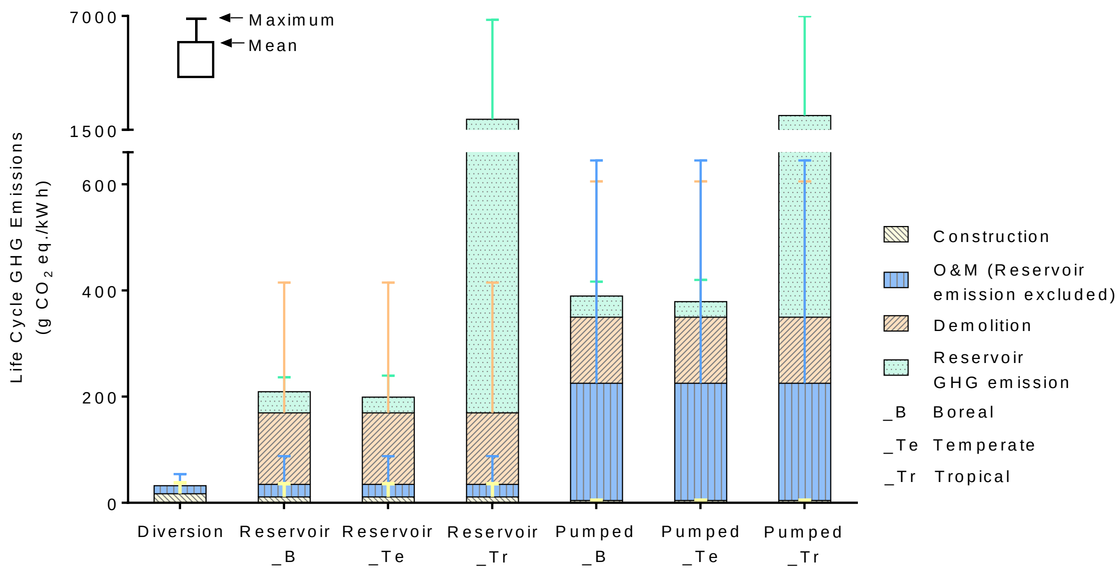


Figure 5-5. Life cycle GHG emissions from dams (The reservoir GHG emissions shown are the global mean values of diffusion, ebullition, and/or degassing emissions from reservoir surface and downstream of dams.)

In order to put the GHG emissions of HPs in perspective, they have been compared with conventional and other renewable electricity-generation technologies and the results are shown in Figure 5-6. River in-stream, run-of-river, and reservoir-based HPs located in boreal and temperate regions generally have a lower GHG emission rate compared with fossil fuel, solar PV, and biomass energy. However, reservoir-based HPs located in tropical regions could have a higher GHG emission rate than fossil fuel energy. Given the importance of reservoir GHG emissions for tropical dams and the potential influence of the GWP characterization time scale on the GHG emissions, a comparison of the 100-year and 20-year GWP was performed for the reservoir GHG emissions (Table D2 of the Appendix D). This comparison was not conducted for other life cycle phases due to a lack of data on the emitted GHG compositions. The GHG emissions per kWh from reservoirs can be up to 2.4 times greater when the 100-year GWP is converted to the 20-year GWP, which further elevates the potential impacts of tropical reservoir-based dams. The 20-year GWP of boreal and temperate dams is around 7-97 and 6-107 g CO_2 eq./kWh respectively, which is still lower compared to the coal-fired (O. Edenhofer et al., 2011) (1,000 g CO_2 eq./kWh) and natural gas (O. Edenhofer et al., 2011) (470 g CO_2 eq./kWh) power generation.

Although reservoir-based HPs located in the tropical regions are shown to have the largest GHG emissions, caution should be exercised in drawing strong conclusions from this comparison due to the uncertainties in the assessment and the specific conditions under which individual projects are evaluated (Pascale et al., 2011). In addition, previous LCA studies only calculated and weighted GHG emissions based on the amount of hydropower generated, while other services provided by dams (e.g., water supply, irrigation, flood control, erosion control, fishing and fire protection) are largely neglected. Furthermore, dams also present environmental impacts other than GHG emissions, such as blocking fish passage, altering natural flow variation, and eliminating small floods and sediment that replenishes stream beds and floodplain soils. These disadvantages should not be neglected. For example, according to Goralczyk's study, hydropower has a light burden for GHG emissions (4.6 g CO_2 eq./kWh) compared with photovoltaic (104 g CO_2 eq./kWh) and wind turbines (6 g CO_2 eq./kWh), but its acidification potential is larger than these two technologies (Góralczyk, 2003). Thus a range of key indicators must be considered when evaluating the sustainability of energy generation technologies

(Turconi et al., 2013). The comprehensive evaluation of the pros and cons of hydropower generation is imperative in decision-making about dam construction, operation, and end-of-life.

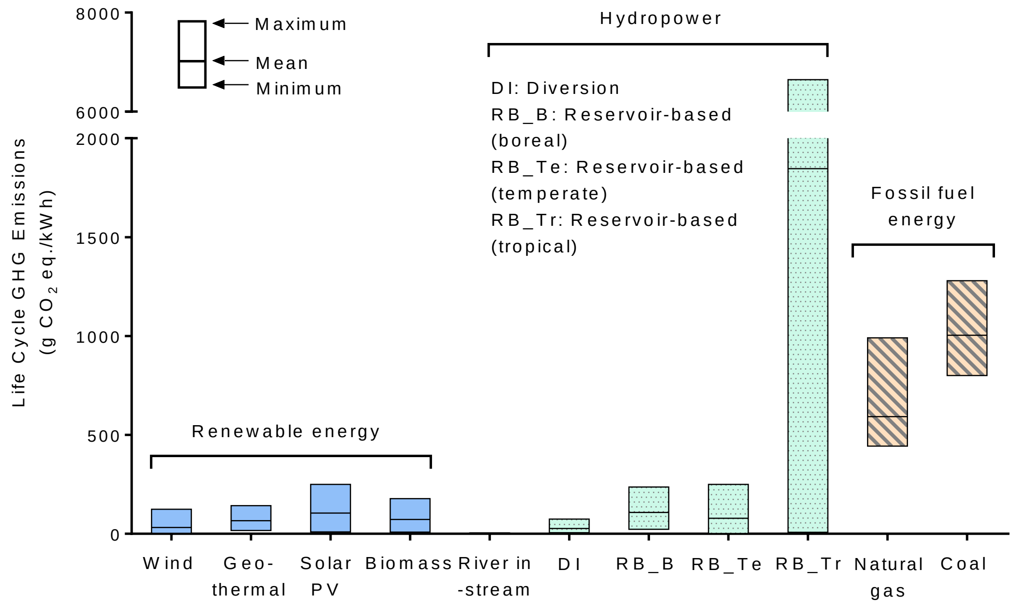


Figure 5-6. Life cycle GHG emissions from different types of energy (*source*: Coal (Dones et al., 2003; Hondo, 2005; Meier et al., 2005; White and Kulcinski, 2000), natural gas (Dones et al., 2003; Meier et al., 2005), wind (Jungbluth et al., 2005; Lenzen and Munksgaard, 2002; Lenzen and Wachsmann, 2004; Schleisner, 2000; White and Kulcinski, 2000), biomass (Carpentieri et al., 2005; Chevalier and Meunier, 2005; Pehnt, 2006), solar PV (Hondo, 2005; Kannan et al., 2006; Tripanagnostopoulos et al., 2005), geothermal (Pehnt, 2006), river in-stream (Gallagher, John et al., 2015; Miller et al., 2011), diversion HPs (Gallagher, J. et al., 2015a; Pang et al., 2015; Varun et al., 2012), reservoir-based (boreal) HPs (Li, S. et al., 2015), reservoir-based (temperate) HPs (Li, S. et al., 2015; Pacca, 2007; Zhang et al., 2007; Zhang, S. et al., 2015), reservoir-based (tropical) HPs (Demarty and Bastien, 2011; Fearnside, 2015a; Li, S. et al., 2015))

5.9. Conclusions

Life cycle GHG emissions from dams are highly site-specific based on different types, scales, and locations of projects. The results of this study considered data from hydropower LCA studies and non-LCA reservoir GHG emission studies. By comparison, published LCA studies estimate a range of 0.2-185 g CO_2 eq./kWh, up to 36 times less than our results. This difference reveals the importance of utilizing a consistent and comprehensive system boundary and considering different dam characteristics in understanding the sustainability of HPs. In general, river in-stream and diversion HPs have much lower GHG emissions compared with reservoir-based HPs. Flooded biomass decomposition, although not commonly considered in existing dam LCAs, is one of the greatest contributors to the GHG emissions of reservoir-based HPs, especially to those located in tropical regions. A comparison among hydro, wind, solar, geothermal, biomass-based, and fossil-fuel-based electricity shows that hydropower generally has comparable GHG emission rates to other types of renewable energy (within a range of 3-250 g CO_2 eq./kWh), but electricity produced from tropical reservoir-based dams could potentially have 27 times higher emission rates than other hydropower and renewables, and around 6 times that of fossil-fuel-based electricity. Collectively, these findings suggest that reservoir-based HPs are viable as a lower GHG emission replacement for fossil-fuel-based electricity in temperate and boreal regions, and river in-stream and diversion HPs are viable options in general. Tropical reservoir-based hydropower is likely to contribute more to climate change than natural-gas-based electricity and possibly even more than coal-based electricity. Hence, decisions regarding new development of hydropower in tropical regions should be made carefully, and should take into consideration the possibility of integrating design measures to minimize GHG production. More studies on the accurate quantification of reservoir GHG emissions are still needed given its potential significance and variability. This study also underscores the need to take a more local/regional approach to energy policy. For example, in a region with site-specific conditions that make reservoir-based hydropower on the higher end of life cycle GHG emissions but biomass or geothermal on the lower end, it may be worthwhile to consider providing greater incentive for the lower-emitting renewable options through carve-outs in a renewable portfolio standard, rather than incentivizing all renewable energy at the same level.

While existing LCAs are primarily focused on hydroelectric dams, the current analysis of NID data revealed potentially equal contribution of reservoir GHG emissions by all non-powered dams (27.03 Tg CO_2 eq./yr) in the USA compared with all hydroelectric dams (28.66 Tg CO_2 eq./yr). Non-powered dams are difficult to assess through LCAs because their primary functions (e.g., recreation, flood control) are often difficult to quantify. Nevertheless, these dams present similar types of impacts as hydroelectric dams. Many of them have approached or exceeded their design life, and shifted their primary functions as they are no longer needed or suited for their original purposes. Some of them remain only because they are costly to be removed or upgraded. As preferences for dams and watershed ecosystem services change, society will need to make thousands of decisions about the future of these dams in the coming decades. Given the diverse uses (e.g., hydropower, water supply, recreation) and consequences of dam presence (e.g., effects on climate change, nutrient flux, habitat availability, diadromous fish populations, safety and liability risks associated with aging infrastructure), alternative decisions for individual dams or networks of dams have unique and emergent economic, technological, environmental, social, and political trade-offs. Multi-scale, integrated social and biophysical analyses are required to provide a holistic view of these trade-offs and to guide future decision-making about dams. The current review is just one of the first steps in quantifying and understanding some of these tradeoffs through the lens of lifecycle GHG emissions. Consideration of future changes in water availability, climate, population, and land use also calls for an improved understanding of their effects on dam operation and management.

CHAPTER 6: ADVANCING SUSTAINABLE MANAGEMENT OF DAMS THROUGH PARTICIPATORY SYSTEM DYNAMICS MODELING

6.1. Introduction

Dams have been heavily constructed worldwide for providing various services such as generating hydroelectricity, minimizing flood risk, creating recreational areas, and providing water for irrigation, towns, and cities. In the United States alone, there are more than 90,000 dams that higher than 1.8 meters (USACE, 2016) without counting over 2 million small low-head ones (Fencl et al., 2015). Most of these dams were built with the emphasis on maximizing services and economic returns from the use of waters, whereas, dams' long-term environmental and social impacts were usually excluded in decision-making because of little or no understanding of that. Over the past 40 to 50 years, there is an increasing awareness that dams alter natural flow regimes, change sediment transportation, degrade water quality, and block aquatic species migration in both obvious and subtle ways which further resulted in the loss of natural resources (e.g., migratory fisheries) and processes (e.g., flood recession cultivation system) that contribute to the livelihoods and well-being of people (Postel and Richter, 2012). In response, dam management plans are often made by comprehensively incorporating and evaluating all aspects of dams (Loucks and Van Beek, 2017). A list of federal and state regulations such as the National Dam Safety Program Act, the Endangered Species Act, the Clean Water Act have been enacted to ensure dam operation and management environmentally sustainable, economically viable, and socially equitable. However, this master planning exercise is usually dominated by professionals in governmental agencies or dam owners without consideration of the concerns and objectives of affected stakeholders, which has proved with little chance to success even if such plan is technically sound.

Participatory process, involving interested stakeholders, concerned citizens, nongovernmental organizations, and professional in governmental agencies in the decision-making process at the outset, is increasingly required for dam sustainable development to give considerations of environmental, economic, and social impacts as well as issues of equity and the rights of people who might be affected positively and adversely (Loucks and Van Beek, 2017; Máñez et al., 2007; McCartney, 2007). Participatory decision-making is controversial not only because of complex dam tradeoffs under different dam management alternatives but also because various or conflict objectives, interests, and agendas of involved stakeholders. In this process, model is recognized as an effective and efficient tool to analyze system performances, facilitate communication and negotiation, and assist stakeholders and decision-makers in reaching a common understanding and agreement (Loucks and Van Beek, 2017). Various types of models (e.g., HEC-ResSim as simulation technique (Klipsch and Hurst, 2007), dynamic programming as optimization technique (Georgakakos et al., 1997), multi-criteria analysis (Said, 2006), Bayesian Networks (Said, 2006)) have been developed and used in either individual or basin-scale dam decision-making with the level of stakeholder engagement in model development changes from 'thin' (minimal involvement, but still providing some feedback on the model after the experts have designed it) to 'thick' (involvement in model design from the beginning, selecting variables, etc.). Despite an increased understanding about tradeoffs arising from different operating systems and an expanded involvement of stakeholders in decision-making, there are still insufficient attempts to evaluate differences between negotiated decision and scientifically optimal alternatives as well as analyze compromises each stakeholder made. In order to fill this knowledge gap, this study applied the energy-fish-cost model in dam negotiation. Two hypotheses were tested in this study: (1) negotiation process might be helpful for assisting stakeholders and decision-makers reaching more balanced outcomes, (2) the extent of compromise each stakeholder made is closely related to their primary interests and objectives.

6.2. Materials and Methods

6.2.1. Proof of concept

Settings of the Pearl River basin

A fictional, costal river basin, named the Pearl River basin, was used as proof of concept. This fictional river mimics a real river located in the New England region U.S. and drains an area of approximately 518 km^2. From its rural and sparsely developed headwaters, the Pearl River flows southeast until it becomes tidal below Dam 1 (Figure 6-1). Historically, the Pearl River basin is home to four sea-run fish species: alewife (*Alosa pseudoharengus*), American shad (*Alosa sapidissima*), Atlantic salmon (*Salmo salar*), and sea lamprey (*Petromyzon marinus*), which provide important recreational, commercial, and ecological values to the local communities. Populations of these fish species have experienced pronounced declines over the past few decades largely due to habitat degradation and dam construction as dams significantly block fish upstream migration and reach to upstream habitat areas. There are five dams in the basin with three hydropower dams on the main stem (Dam-1~3) and two non-hydropower dams (Dam-A and B) on the Mill Creek, a tributary of the Pearl River. As part of their last federal relicensing process requirement, three hydropower dams have installed specific fish passage structures (hereafter called fishways).

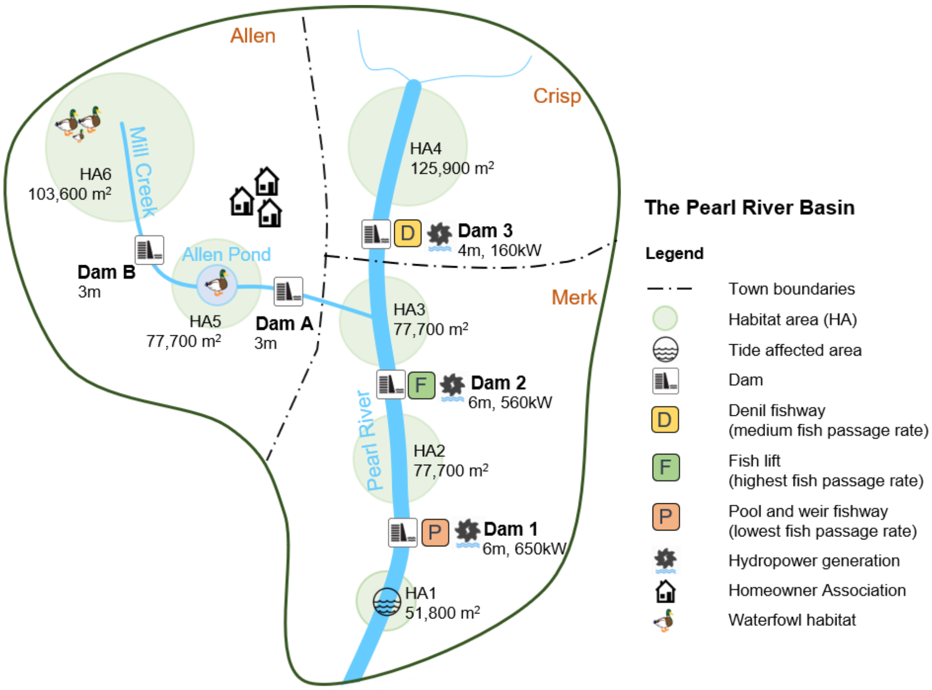


Figure 6-1. Map of the Pearl River basin showing the siting and current status of the five dams. The whole basin was divided into six habitat areas according to the location of these dams.

Characteristics of the five dams are provided in Table 6-1. We designated siting of dams throughout the Pearl River as well as drainage area above each dam site. Drainage area was used to estimate the bankfull channel geometry (e.g., width, mean depth, and cross-sectional area) and bankfull discharge according to regression equations provided in (Bent and Waite, 2013) (refer the Appendix E for detailed information). Bankfull discharge is the streamflow that fills the cross section without overtopping the banks. The recurrence intervals of bankfull discharges for 77 study sites range from 1.03 to 3.48 years roughly corresponding to the flow with an exceedance fraction of 10~15% (Bent and Waite, 2013; Naito and Parker, 2016, 2019). Installed or potential hydropower capacity at each dam site was estimated based on design discharge (or turbine release capacity) and available head (or turbine rated head, Equation 6-1). For run-of-river power plants, the recommendation is to choose a design discharge that is available 100 to 120 days a year or has an exceedance fraction of 30% (Giesecke et al., 2014). This suggested level of design discharge is roughly 30% of bankfull discharge according to historical flow data at the Trinity River near Romayor, TX and the Minnesota River, MN (Naito and Parker, 2016, 2019). Hence, turbine release capacity at each site was assumed to be 30% of bankfull discharge. Turbine rated head was assumed to be 3 times larger than its bankfull mean depth considering that the best dam location is usually at the locations where there are narrowing of the river.

$P = Q \times H \times \eta \times \rho \times \mathrm{g} \times 10^{-6}$ Equation 6-1

where P is the installed hydropower capacity (or theoretical power produced at the transformer), MW; Q is turbine release capacity, m^3/s; H is the rated head, meters; η is the overall efficiency, assumed to be 0.85 (Hadjerioua et al., 2012; Power, 2015); ρ is the density of water, 1,000 kg/m^3; and, g is the acceleration due to gravity, 9.8 m/s^2.

Table 6-1. Project information of the five dams in the Pearl River basin.

Dams	Owner	Primary purpose	Drainage area* (km^2)	Dam height (m)	Turbine release capacity (m^3/s)	Rated head (m)	Installed /potential capacity (kW)	Installed fishway
Dam-*1*	Hydro Energy, LLC	Hydroelectric	466	6	20	3.9	650	Pool-and-weir
Dam-*2*	Hydro Energy, LLC	Hydroelectric	389	6	17	3.9	560	Fish lift
Dam-*3*	Hydro Energy, LLC	Hydroelectric	130	4	7	2.7	160	Denil
Dam-*A*	Town of Allen	Recreation	181	3	*9*	*1.8*	*135 (potential capacity)*	None
Dam-*B*	Town of Allen	Recreation	104	3	*6*	*1.8*	*90 (potential capacity)*	None

Note:
*Drainage area is defined as the land area where precipitation falls off into creeks, streams, rivers, lakes, and reservoirs above each dam site (USGS, 2019).

Settings of upcoming dam negotiations

The setting of the Pearl River role-play negotiation simulation used in this study (Diessner et al., 2020) consists of a diverse set of interests, issues, and stakeholders. The Town of Allen was recently issued a notice of deficiently, which indicated that Dam-*A* poses a threat to public safety and that deficiencies must be addressed if the Town is to comply with state regulations. The notice is particularly controversial among the Town's residents because there are other stakeholders who are interested in seeing the dam being removed for ecological benefits. Additionally, the timing of this notice of deficiency coincides with an upcoming relicensing process for the hydropower dams on the Pearl River (Dams 1-3). The state WRD believes that this set of multiple and diverse issues provides interested stakeholders with the opportunity to discuss and plan for the future of the Pearl River basin. The WRD organized a meeting of the parties to discuss these opportunities and agree on a Work Plan to move forward. Seven interested parties (or roles) who influence and be influenced by dam management decisions were included in the Working Group: Federal Agency of Natural Resources, State Water Resources Division, Historic Preservation Agency of the State, HydroEnergy, LLC., Allen Pond Homeowner Association, Rivers-R-Us, Town of Allen Municipal Official (Table 2) (Diessner et al., 2020). The interests and concerns of each role were identified based upon data from stakeholder interviews (Diessner and Ashcraft, n.d.). As part of the Work Plan, stakeholder needed to reach agreement on three critical decisions: (1) which dams should be included in the Work Plan and what dam management alternatives should be considered? (2) Who is responsible for implementing the Work Plan? (3) Who pays to implement the Work Plan? To support decisions about how to manage dams, stakeholders engaged in dam negotiation are agreed to use a simulation tool that simulates hydropower generation, economic cost, and population of four native sea-run fish species under different dam management alternatives. This paper is mainly focused on introducing the development of this simulation tool and discussing the negotiated results related to the first decision.

Table 6-2. The designed roles engaged in dam negotiation and their main interests.

Role	Type of stakeholder	Interests

Federal Agency of Natural Resources (FANR)	Federal government	Improve fish populations Improve ecosystem health and resilience (e.g. open up river miles, improve upstream habitat quality) Participatory decision-making and community support for proposed projects
State Water Resources Division (WRD)	State government	Safety improvements Improve fish populations Improve ecosystem health & resilience (e.g. open up river miles, improve upstream habitat quality)
Historic Preservation Agency of the State (HPAS)	State government	Preservation of historic resources Participatory and transparent decision-making (specifically early involvement of historic interests in the process)
HydroEnergy, LLC.	Hydropower developer and operator; Dam owner	Hydroelectricity generation Reduce uncertainty & costs related to the upcoming relicensing process
Allen Pond Homeowner Association (HOA)	Property owners along Allen Pond	Maintain property values Maintain or improve pond-based recreation Maintain thriving waterfowl habitat
Rivers-R-Us	Non-governmental, non-profit organization	Improve fish populations Improve ecosystem health and resilience (e.g. open up river miles, improve upstream habitat quality) Improve river-based recreation
Town of Allen Municipal Official (Town)	Municipal government; Dam owner	Safety improvements related to Dam A Foster economic vitality Participatory and transparent decision-making
Facilitation Team	Neutral third-party; Not a stakeholder	N/A

6.2.2. Development of a simulation tool: a collaborative stakeholder-informed approach

The development of a simulation tool includes two stages: building of system dynamics model and developing a user-friendly web application. Both stages were conducted in close collaboration with experts and stakeholders.

Building of system dynamics model

An integrated SDM model, consisting of an age-structured fish population model, an energy model, and a cost model, was built in Vensim® DSS on a daily time step. This model was adapted from an extant model in Song et al. (2020). Six system environmental and economic performance metrics that affect and be affected by dam decision-makings were modeled: spawner population potential of four sea-run fish species (number of spawners), annual hydropower generation (GWh/year), and project cost ($ million). These system performance indicators were chosen as they reflect major interests of a wide array of stakeholders associated with dam management (Diessner et al., 2020). Additionally, the quantitative nature of these metrics and reliable data availability for each metrics make them desirable candidates for system dynamics modeling.

The age-structured fish population model was built in collaboration with fishery scientists. It simulates spawner population potential of four fish species in the freshwater that are ready to spawn each year by keeping track of their growth, mortality, maturity, iteroparity, timing, period, and routes of migration at each life stage throughout the whole life span. The stabilized fish population potential was used in analysis by running the model 150 years.

The energy model simulates theoretical hydropower generation each year at all hydropower dams (GWh/y). It was calculated as a product of the installed hydropower capacity (Equation 1) and annual turbine operation period (hours) (Adeva Bustos et al., 2017; Hadjerioua et al., 2012; Singh and Singal, 2017). Daily turbine operation period equals to 24h or 0 if turbine shutdown is operated.

The cost model calculates total project costs related to dam repair, removal, and installation of fishway and hydropower from all five dams. Dam repair cost is solely applicable to Dam-*A*, the one that has received the notice of deficiency. Dam-*A* repair cost was assumed to $0.5 million. Dam removal cost was simulated by multiplying the dam height with the average dam removal cost per vertical meter rise of the dam height, $ 0.384 million/meter, which was calculated based upon removal cost from (Maclin and Sicchio, 1999) and 37 removal projects in the New England region. Fishway installation cost considered capital investment along with operation and maintenance (O&M) cost over a 30-year planning horizon. This time period is consistent with

FERC license period for non-federal owned hydroelectric dams (Madani, 2011). Capital investment of fishway installation was predicted as a product of the dam height and the unit capital cost per vertical meter rise of the dam height (refer Table XX in the SI for detailed information). Annual O&M cost was estimated to be 2% of the capital cost (Nieminen et al., 2017). Turbine installation cost is applicable to Dam-*A* and Dam-*B*, which calculates cost of constructing a hydropower plant on non-powered dams by multiplying potential hydropower capacity at each site and average unit cost per hydropower capacity ($5,000/kW) (O'Connor et al., 2015). The detailed relationships, equations, parameter values, and assumptions associated with energy-fish-cost model are provided in Section B of the SI.

Dam management alternatives. Five groups of dam management strategies were considered: (1) no action or repair (only for Dam-*A*), (2) install one fishway, (3) install two different types of fishways, (4) install hydropower (only for Dam-*A* and Dam-*B*), and (5) removal. Given that the effectiveness of fishways in facilitating fish upstream passage varies markedly based upon the types and numbers of fishway installed as well as the types of fish species (Bunt et al., 2012; Noonan et al., 2012). We studied four widely applied fishways: pool-and-weir fishway, Denil fishway, fish lift, and nature-like fishway. For three hydropower dams on the mainstem (Dam-*1~3*), installation of nature-like fishway is not an applicable option as it is uncommon to install nature-like fishway at higher dams. Thus, there are 4 different dam management options for each of the three hydropower dams (Table 2). For Dam-*A* and Dam-*B*, dam management options of either install hydropower or not could further be applied to three groups of dam management strategies: (1) no action or repair (only for Dam-*A*), (2) install one fishway, (3) install two different types. In total, there are 23 management options at both Dam-*A* and Dam-*B* (Table 3).

To provide a complete picture of the accumulated effects of multiple dams, the model captured all possible permutations of the studied five dams ($4^3 + 23^2 = 33{,}856$ alternatives).

Table 6-3. The designed dam management options at each dam

Dams	Manage options at each dam				#Options
Dam-*1*	No action	Install Denil	Install fish lift	Removal	4
Dam-*2*	No action	Install Pool-and-weir	Install Denil	Removal	4
Dam-*3*	No action	Install Pool-and-weir	Install fish lift	Removal	4
Dam-*A*	Repair	Install 1 out of four fishways	Install 2 out of four fishways	Removal	23
		(These options may either install hydropower or not)			
Dam-*B*	No action	Install 1 out of four fishways	Install 2 out of four fishways	Removal	23
		(These options may either install hydropower or not)			

Development of web application

Given the complexity of integrated SDM model, a user-friendly web application was developed to present system performances under different dam management alternatives in close cooperation with software engineers. This web application includes a table page controlling management options at each dam and a result page showing associated outcomes of the six system performance indicators (visit http://dam.gsscdev.com/dam-system-dynamics/ for more details). The design and characteristics of the web application were initially made by co-authors and further refined based on feedback from workshop participants, including discipline-specific experts and stakeholders.

6.2.3. Performance measures, calculation, and illustration

Due to inconsistency in units and preferred performances of the six system performance indicators, a normalized value indicating the extent of preference was adopted for analysis (Equation 6-2).

$$N_{i,j} = \frac{P_{i,j} - L_i}{M_i - L_i} \qquad \text{Equation 6-2}$$

where $N_{i,j}$ is the normalized value of system performance indicator i for dam management alternative j (j =1~33,856) with 0 meaning the least preferred situation and 1 meaning the most preferred situation. $P_{i,j}$ is numerical value of system performance indicator i under dam management alternative *j*. L_i and M_i are the least and the most preferred numerical value of system performance indicator *i* in the Pearl River system, respectively.

In optimizing fish-energy-cost problems, the goodness of a dam management alternative is determined by the dominance. Alternative $\boldsymbol{x_1}$ dominates alternative $\boldsymbol{x_2}$ (or $\boldsymbol{x_2}$ is dominated by $\boldsymbol{x_1}$) if $\boldsymbol{x_1}$ is no worse than $\boldsymbol{x_2}$ in all objectives, and $\boldsymbol{x_1}$ is strictly better than $\boldsymbol{x_2}$ in at least one objective (Marler and Arora, 2004). Pareto-optimal solution is a set of all solutions that are not dominated by any member of the solution set. The boundary defined by the set of all points mapped from the Pareto-optimal set is called Pareto-optimal front. Pareto-optimal dam management alternatives were determined using the *nondominated_points()* function from the 'emoa' package in R. Parallel coordinate plot and radar chart were adopted to illustrate system environmental performances under various dam management alternatives. These two types of visualization techniques are ideal for comparing multiple variables and analyzing the relationships between them (Siirtola et al., 2009). Parallel coordinate plot was plotted using the *ggparcoord()* function form the 'GGally' package in R. Radar chart was plotted using the *radarchart()* function form the 'fmsb' package in R.

6.3. Results and Discussion

6.3.1. Performances of fish-energy-cost tradeoffs

The minimum and maximum numerical values of the six studied system performance indicators in the Pearl River system are (0, 12.3) for hydropower generation (GWh/y), (0, 11.2) for project cost ($ million), and (52.4, 199.3), (2.7, 11.1), (2.8, 4.5), (24.4, 102.6) for spawner population potential of Alewife, American shad, Atlantic salmon, sea lamprey are, respectively. The least preferred numerical value of project cost is its maximum numerical value, while the least preferred values of other indicators are their minimum numerical values. The normalized values of these system performance indicators under all possible dam management alternatives were calculated and shown in Figure 6-2.

Under the baseline condition (Baseline, black polyline in Figure 6-2), the normalized value of energy generation is 0.84 (or 84% of its maximum value) as no power generated from Dam-*A* and Dam-*B*. Meanwhile, the normalized value of project cost and spawner population potential of four fish species are zero or the least preferred situation. Except for the baseline condition, the remaining dam management alternatives were divided into two groups: Pareto-optimal alternatives and alternatives that are not Pareto-optimal alternatives (also known as suboptimal alternatives, gray polylines in Figure 6-2). According to the results of non-dominated point analysis, 661 out of 33,856 dam management alternatives were identified as Pareto-optimal alternatives. These Pareto-optimal alternatives defines a set of solutions that none of the six system performance indicators can be improved in value by any other feasible alternatives without worsening at least another indicator value (Almeida et al., 2019; Roy et al., 2018). Based upon performances of the six system indicators, we further divided Pareto-optimal alternatives into five groups: alternatives with the most preferred situation of (1) energy generation (Pareto_MaxE, green polylines), (2) project cost (Pareto_MinC, red polyline), and (3) spawner population of four fish species (Pareto_MaxF, blue polyline), (4) alternatives that balance fish-energy-cost tradeoffs (Pareto_Balanced, purple polylines), and (5) other Pareto-optimal alternatives (Pareto_Others, yellow polylines).

Strong conflicts between energy generation, project cost, and fish populations exist when maximizing either one of the six system indicators. For example, the goal of generating the highest energy from the river basin (Pareto_MaxE in Figure 6-2) can be achieved if both Dam-*A* and Dam-*B* installed hydropower along with keeping the remaining three hydropower dams on the mainstem. Project cost changes in the range of 15% to 87% of its maximum value, while Alewife, American shad, Atlantic salmon, and sea lamprey changes in the range of 26~73%, 24~71%, 62~95%, and 24~56% respectively. Specifically, strong conflicts between fish and cost exist as high fish population potential (preferred situation) lead to high project cost (nonpreferred situation) and vice versa. The minimum project cost alternative is associated with only repairing Dam A (Pareto_MinC in Figure 6-2). However, population of four species are at their least preferred level. When fully restoring four types of fish species by removing all five dams (Pareto_MaxF in Figure 2), energy generation is at its lowest value and project cost is relatively high with around 75% of its maximum value. Therefore, it is impossible to simultaneously maximize six system performance indicators due to conflict nature between them.

In order to achieve solutions that may balance energy generation, fish populations, and project cost, corresponding compromises should be made. When energy generation is larger than 60% of its maximum value, spawner population potential of Alewife, American shad, Atlantic salmon, and sea lamprey is no higher than 44%, 42%, 78%, and 39% of their maximum values if lowering project cost to 20% of its maximum value. If limiting project cost to 40% of its maximum value, populations of the corresponding four fish species are potentially increased to 61%, 60%, 80%, and 57% of their maximum values. Keeping the limitation of energy generation unchanged, spawner population potential of these four fish species may maximize to 86%, 86%, 95%, and 84% of their maximum values while increasing project cost to 85% of its maximum value. Based upon these results, we define Pareto-optimal alternatives that meet the following three criteria as alternatives balancing fish-energy-cost tradeoffs. (1) Energy generation is larger than 60% of the maximum hydropower potential, (2) project cost is less than 40% of the maximum value, (3) populations of four types of fish species are larger than 50% of their maximum populations that the river can support. Six Pareto-optimal alternative were identified as solutions that balance fish-energy-cost tradeoffs which maximize energy generation, alewife, American shad, Atlantic salmon, and sea lamprey to 84~94%, 54~59%, 52~58%, 78~99%, and 51~56% of their maximum values, while minimizing project cost to 32~39% of its maximum value (Pareto_Balanced in Figure 2). These balanced Pareto-optimal alternatives are associated with installing Denil fishway at Dam-1, installing Denil or pool-and-wire fishway at Dam-2, doing nothing at Dam-3, removing Dam-A, doing nothing, or installing nature-like fishway or installing hydropower at Dam-B. In addition, balanced Pareto-optimal alternative may also be achieved through installing Denil fishway at Dam-1 and Dam-2, doing nothing at Dam-3, repairing and installing nature-like fishway and hydropower at Dam-A, and doing nothing at Dam-B.

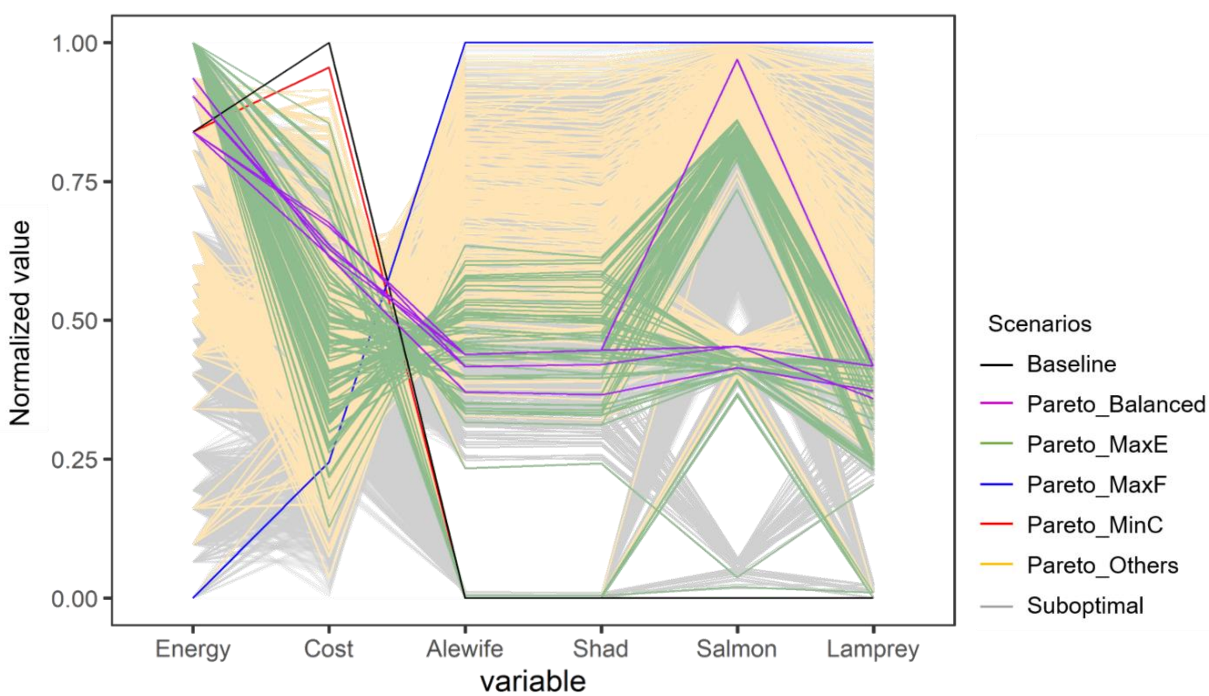


Figure 6-2. Parallel coordinate plot illustrates tradeoffs among hydropower generation, project cost, and fish population potential of four sea-run fish species under all possible dam management scenarios in the Pearl River basin. Each vertical axis represents performance of the six objectives. Each polyline represents one out of the 33,856 dam management alternatives. Each alternative's performance is designated by the points at which it intersects each vertical axis. The steepness of the diagonal lines

between two adjacent axes displays the degree of conflict between the two objectives. Normalized values: 0 indicates the least preferred scenario and 1 indicates the most preferred scenario

6.3.2. Balanced Pareto-optimal alternatives vs. negotiated decisions

Dam management decisions listed in Table 6-4 are negotiated decisions made by four stakeholder negotiating groups at two different workshops. The normalized values of six performance indicators under all negotiated decisions are provided in Table 6-4 and Figure 6-3(A). The results shown that all four negotiated decisions are consistent in preserving high level of energy generation. This might be explained by the fact that dam removal is difficult under the constraints present in real life decision-making. Among all four negotiated decisions, NH-1 decision has the least preferred level of salmon and NH-2 decision presents the lowest preferred condition on project cost. Performances of negotiated decisions made at NH-3 and RI-1 workshop groups are similar except that NH-3 decision has a higher population of salmon and slightly lower energy generation.

Table 6-4. Negotiated decisions made by four stakeholder negotiating groups at two different workshops

Groups	Negotiated dam management decision					Normalized values of six performance indicators					
	Dam-1	Dam-2	Dam-3	Dam-A	Dam-B	Energy	Cost	Alewife	Shad	Salmon	Sea lamprey
NH-1	Install fish lift	No action	No action	Repair	Remove	0.84	0.66	0.29	0.29	0.04	0.20
NH-2	Install Denil fishway	No action	Remove	Repair, install HP and nature-like fishway	Install HP and nature-like fishway	0.90	0.50	0.38	0.37	0.75	0.26
NH-3	Install Denil fishway	No action	No action	Repair, install nature-like fishway	Remove	0.84	0.69	0.33	0.32	0.87	0.24
RI-1	Install Denil fishway	No action	No action	Repair, install HP and nature-like fishway	No action	0.94	0.73	0.32	0.31	0.40	0.24

A comparison between Figure 6-3 (A) and (B) shows that the NH-3 and RI-1 negotiated decisions present similar performances on fish-energy-cost tradeoffs compared with the balanced Pareto-optimal alternatives. In order to measure the differences between negotiated decisions and the balanced Pareto-optimal alternatives, the extent of gain/loss of each negotiated decision was calculated using Equation 6-3.

$T_n = \sum_{i=1}^{6} w_i \times (N_{i,n} - N_{i,b})$ Equation 6-3

where T_n is the extent of gain/loss of negotiated decision n. n indicates four negotiated decisions made at four workshop groups. i refers six system performance indicators (i = 1~6). w_i is weighting coefficient of six system performance indicators ($w_i > 0$). $N_{i,n}$ and $N_{i,b}$ are the normalized values of system performance indicator i under negotiated decision n and balanced Pareto-optimal alternatives, respectively. If the value of T_n is positive, negotiated decision n gains compared with balanced alternative. Otherwise, negotiated decision n losses. The relative value of T_n is closer to zero meaning negotiated decision n is better in performing more balanced outcomes.

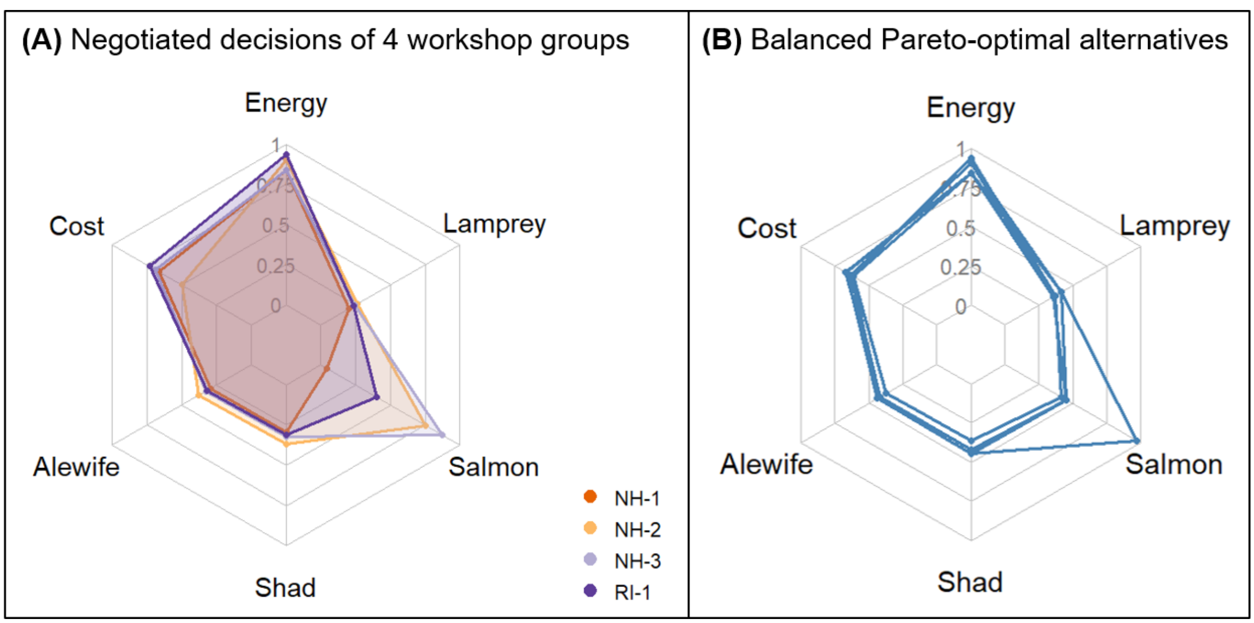


Figure 6-3. Radar chart presenting performances of energy, cost, and populations of four fish species under (A) balanced Pareto-optimal alternatives and (B) negotiated decisions at four workshop groups.

The extent of gain/loss was analyzed under two weighting coefficient scenarios (Table 6-5). The first scenario was related to equally weight all six performance indicators where $w_i = 1$ ($i = 1{\sim}6$). The second scenario was seeing four types of fish species as a group which was then equally weighted with energy generation and project cost ($w_i = 1$ for energy and cost indicators, $w_i = 0.25$ for fish indicators). The results shown that when all six performance indicators equally valued, all negotiated decisions loss compared with balanced Pareto-optimal scenarios. This is because the summed loss of four types of fish species outweigh potential gains from energy and cost for all negotiated decisions. The results also show that the negotiated decisions are suboptimal solutions and more than one of the six objectives can be improved through other feasible solutions. When applying the second weighting scenarios, the NH-1 and NH-2 decisions loss, whereas, the NH-3 and RI-1 decisions gain. The gaining of the NH-3 and RI-1 decisions stems from outperforming in either energy or cost. No matter which weighting scenarios were applied, the NH-3 decision performs the closest outcomes in comparison with the balanced Pareto-optimal alternatives.

Table 6-5. Average extent of gain/loss of four negotiated decisions compared with balanced Pareto-optimal alternatives

Negotiated groups	Average extent of gain/loss (w_i = 1 for all six system indicators)	Average extent of gain/loss (w_i = 1 for energy and cost indicators, w_i = 0.25 for fish indicators)
NH-1	-0.95	-0.25
NH-2	-0.11	-0.11
NH-3	-0.06	0.01
RI-1	-0.35	0.02

6.3.3. The extent of gain/loss for each designed role

The extent of gain/loss each stakeholder achieved/compromised at all four negotiated decisions was analyzed according to stakeholders' minimally acceptable alternatives (Equation

6-4). This value also investigated based upon stakeholders preferred alternatives which are provided in Section E2 of the Appendix E.

$T_{r,n} = \sum_{i=1}^{6} w_i \times (N_{i,n} - N_{i,r})$ Equation 6-4

where $T_{r,n}$ is the extent of gain/loss stakeholder r achieved under negotiated decision n. r denotes seven designed roles attending workshop groups and n indicates four negotiated decisions made at four workshop groups. Refer Equation 6-3 for the meaning of parameter i, w_i and $N_{i,n}$. $N_{i,r}$ is the normalized values of system performance indicator i under minimally acceptable alternatives of stakeholder r.

According to the primary interests and constraints of each designed role, their minimally acceptable alternatives were identified and listed in Table 6-6. For simplification and illustration, we only analyzed one minimally acceptable alternative for each role. It should be noted that each role may have more than one minimally acceptable alternatives because similar outcomes could be achieved through different basin-scale dam management options. Additionally, the minimally acceptable alternative could vary from player-to-player, and as long as the player follows the constraints of their assigned role, it is possible to have numerous "minimally acceptable" alternatives.

Table 6-6. Examples of minimally acceptable outcomes for each role based on their [quantifiable] primary interests. Minimally acceptable outcomes are not limited to these options (as there are a variety of minimally acceptable scenarios), although these outcomes represent examples of what might be minimally acceptable for each role.

Role	Dam management alternatives					Normalized values of six performance indicators					
	Dam-1	Dam-2	Dam-3	Dam-A	Dam-B	Energy	Cost	Alewife	Shad	salmon	Sea lamprey
FANR, WRD, Rivers-R-Us	Install Denil	No action	No action	Repair, install Denil	Install Denil	0.84	0.73	0.33	0.32	0.44	0.23
HPAS, Town, HOA	No action	No action	No action	Repair	No action	0.84	0.96	0	0	0	0
Hydro Energy LLC.	No action	No action	No action	No action	No action	0.84	1	0	0	0	0

Under these minimally acceptable dam management alternatives, the normalized values of six performance indicators are provided in both Table 6-6 and Figure 6-4 (A). The results shown that the studied seven roles can be roughly divided into two groups based upon their preference for energy, fish, and cost. One group includes three roles: FANR, WRD, and River-R-Us, who mainly focused on improving fish populations as well as ecosystem health and resilience. Their minimally acceptable management actions listed here is installing Denil fishway at Dam-1 (the most downstream dam at the mainstem), Dam-A, and Dam-B (at the upstream dam at the tributary). Fishway installations at multiple dams lead to significant increase in fish population along with increasing of cost. Another group includes the remaining four roles: HPAS, Town, HOA, and HydroEnergy LLC. For these roles, they mainly pay attention on historical values, property values, pond-based recreation, and safety issues related to Dam-A. Therefore, their preferred management actions are solely about managing (e.g., repair, install turbine and fishway) Dam-A. Although these actions have superior performance in energy generation and project cost, fish population potential of four fish species is relatively low.

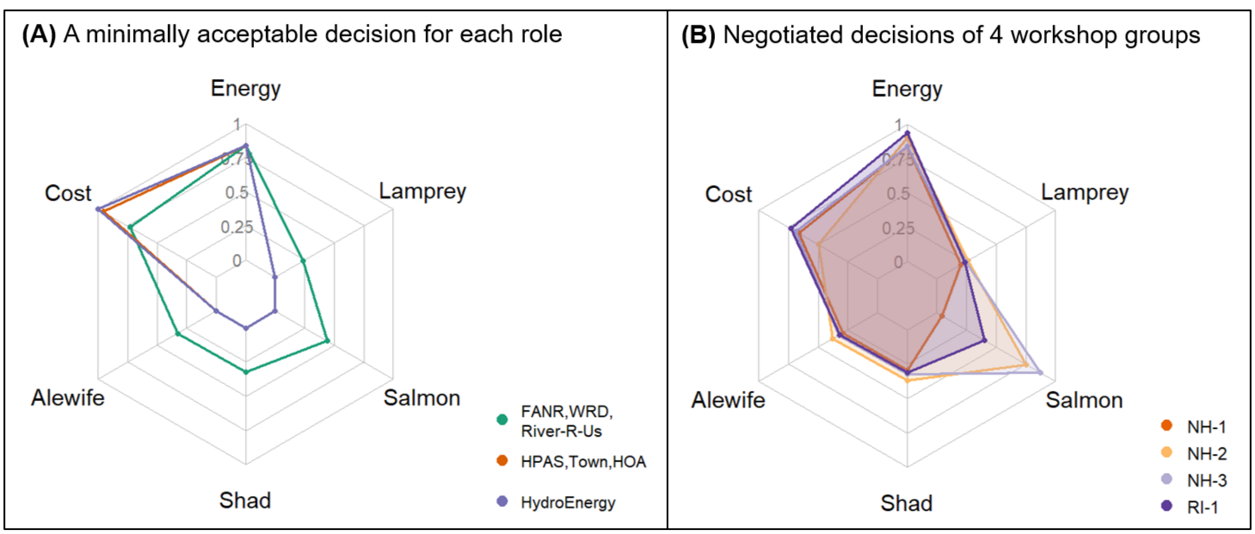


Figure 6-4. Radar chart presenting performances of energy, cost, and populations of four fish species under (A) minimally acceptable dam management scenarios for all seven roles, (B) negotiated decisions at four workshop groups.

For all four negotiated decisions, the quantitative value of extent of gain/loss each stakeholder achieved is provided in Table 6-7. Under the first weighting scenario, compromises were solely made by participants playing the roles of River-R-Us, FANR, and WRD at NH-1 negotiating group because they agreed on negotiated agreement that did not meet their minimally acceptable outcome for salmon population. For the remaining four roles (HPAS, Town, HydroEnergy, and HOA), participants playing these roles always gain at all four workshop groups. For all designed roles, the extent of gain/loss achieved follows sequences: HPAS, Town, and HOA > HydroEnergy > FANR, WRD, and River-R-Us. For all negotiation groups, the extent of gain/loss made by all roles follows the sequence: NH-3 > NH-2 > RI-1 > NH-1. This sequence is consistent with the differences between negotiated decision and balanced Pareto-optimal alternatives. If applying the second weighting scenario, slightly different results were obtained. Except for participants playing the roles of River-R-Us, FANR, and WRD, participants who play the remaining four roles also made compromises at NH-1 negotiation group as they agreed on negotiated agreement that did not meet their minimally acceptable outcome for cost. Interestingly, the sequence of extent of gain/loss for all roles does not change. For all negotiation groups, the extent of gain/loss made by all roles follows the sequence: RI-1 > NH-3 > NH-2 > NH-1, which also keep consistent with differences between negotiated decision and balanced Pareto-optimal alternatives. The results shown that stakeholders gain more in reaching negotiation decisions that are closer to scientifically optimal alternatives.

Table 6-7. Extent of gain/loss of each role under four negotiated decisions based on one of their minimally acceptable alternatives

Role	Extent of gain/loss (w_i = 1 for all six system indicators)	Extent of gain/loss (w_i = 1 for energy and cost indicators, w_i = 0.25 for fish indicators)

	NH-1	NH-2	NH-3	RI-1	NH-1	NH-2	NH-3	RI-1
FANR, WRD, Rivers-R-Us	-0.55	0.28	0.40	0.05	-0.19	-0.05	0.07	0.09
HPAS, Town, HOA	0.54	1.37	1.49	1.14	-0.09	0.05	0.17	0.19
HydroEnergy, LLC.	0.49	1.33	1.45	1.10	-0.13	0.01	0.13	0.14

APPENDIX A: SUPPORTING INFORMATION FOR CHAPTER 2

Section A1. Estimation of daily streamflow at five hydroelectric dams

The drainage-area ratio method is one commonly used method to estimate streamflow for sites where streamflow data were not measured. This method estimates flow at an ungagged location, by multiplying the measured flow at the nearby reference gage by the drainage area ratio of the ungagged to gaged watersheds (Equation A1), with the assumption that the flow per unit drainage area is the same at both the ungagged location and the gaged location (Archfield and Vogel, 2010; Gianfagna et al., 2015):

$$Q_{dam} = Q_{gage} \times \frac{A_{dam}}{A_{gage}} \qquad \text{(Equation A1)}$$

where Q and A represent streamflow and watershed area, respectively.
The detailed processes of streamflow calculation at each dam are provided below.
Mattaceunk Dam
Flows at Mattaceunk Dam were calculated based on the flows reported at West Enfield gage (USGS, 2001-2015), prorated by a factor of 0.502, which represents the drainage area of the project (3349 mi^2) site divided by the drainage area of West Enfield gage (6671 mi^2).
West Enfield Dam
Flows at West Enfield Dam equalled the reported flows at West Enfield gage subtract the reported flows at Medford gage (USGS, 2001-2015) which is a major tributary located at downstream of the project but upstream of West Enfield gage.
Milford Dam and Great Works Dam
Great Works Dam is located at only 2 km downstream of Milford Dam. Thus, we assumed that flows at Milford Dam and Great Works Dam are equal as the close distance and no major tributaries exist between these two dams. Flows at the two dams were calculated based on the reported flows at the West Enfield gage, prorated by a factor of 0.75, which represents the drainage area of the project site (7515 mi^2) divided by the drainage area of West Enfield gage (6671 mi^2) multiplied by 2/3 (the remaining 1/3 flows divert to the Stillwater Brach).
Veazie Dam
Flows at Veazie Dam were calculated based on the flows reported at the West Enfield gage, prorated by a factor of 1.13, which represents the drainage area of the project (7515 mi^2) site divided by the drainage area of the West Enfield gage.

Section A2. Parameters in the age-structured fish population model

The value of input parameters in the fish population model was compiled and calculated based on data published in the literatures and reports. Specifically, the fecundity relationship and other demographics (i.e., mass of each age group) were calculated using St. Croix Milltown alewife trap data.

Table A1. Parameters used in the age-structured fish population model

Parameter	Parameter code	Value or relationship	Source
Average instantaneous natural mortality rate	*M*	0.85	(Atlantic States Maine Fisheries Commission, 2012; Barber et al., 2018)

Average interval spawning mortality rate	M_{spawn}	0.45	(Barber et al., 2018; Durbin et al., 1979; Kissil, 1974)
Instantaneous annual ocean mortality rate	M_{ocean}	0.648	(Barber et al., 2018)
Medium interval fishing mortality	$M_{fishing}$	0.4	(Barber et al., 2018)
Fecundity relationship	F	$y = bx - c$	(Barber et al., 2018; Fisheries and Oceans Canada et al., 1981-2016)
Fecundity slope	b	872	(Barber et al., 2018; Fisheries and Oceans Canada et al., 1981-2016)
Fecundity intercept	c	50916	(Barber et al., 2018; Fisheries and Oceans Canada et al., 1981-2016)
Lifetime reproductive rate	α	0.0015	(Barber et al., 2018)
Asymptotic recruitment level (age-0 fish/acre)	R_{asy}	3283	(Barber et al., 2018)
Maturity between age-2 and age-3	m_3	0.35	(Gibson and Myers, 2003)
Maturity between age-3 and age-4	m_4	0.51	(Gibson and Myers, 2003)
Maturity between age-4 and age-5	m_5	0.96	(Gibson and Myers, 2003)
Maturity between age-5 and age-6	m_6	0.96	(Gibson and Myers, 2003)
Mass of age-3 (g)	W_3	144	(Barber et al., 2018; Fisheries and Oceans Canada et al., 1981-2016)
Mass of age-4 (g)	W_4	186	(Barber et al., 2018; Fisheries and Oceans Canada et al., 1981-2016)
Mass of age-5 (g)	W_5	209	(Barber et al., 2018; Fisheries and Oceans Canada et al., 1981-2016)
Mass of age-6 (g)	W_6	244	(Barber et al., 2018; Fisheries and Oceans Canada et al., 1981-2016)
Sex ratio	$F{:}M$	0.5	(Barber et al., 2018)
Turbine mortality rate	$M_{turbine}$	0.3	(Pracheil et al., 2016)
Fishway passage efficiency	$P_{fishway}$	0.7	(Bunt et al., 2012; Franklin, 2009)
Potential habitat area below Veazie (acre)	$HA1$	7963	(TNC, 2016)
Potential habitat area between Veazie and Great Works (acre)	$HA2$	2379	(TNC, 2016)
Potential habitat area between Great Works and Milford (acre)	$HA3$	0	(TNC, 2016)
Potential habitat area below Milford and West Enfield (acre)	$HA4$	29506	(TNC, 2016)
Potential habitat area between West Enfield and Mattaseunk (acre)	$HA5$	25865	(TNC, 2016)
Potential habitat area above Mattaseunk (acre)	$HA6$	15680	(TNC, 2016)
Total potential habitat area (acre)	HA	81393	(TNC, 2016)

Section A3. Behaviour test of energy model

The correlation coefficients (r^2) between simulated and historical yearly hydropower generation at Milford and West Enfield dams in the period of 2001 to 2015 are provided in Figure A1 (a) and (b), respectively. According to the results, the calibrated r^2 at Milford Dam and West Enfield Dam are 0.60 and 0.86, respectively. The goodness of fit at both sites are deemed reasonable for this study.

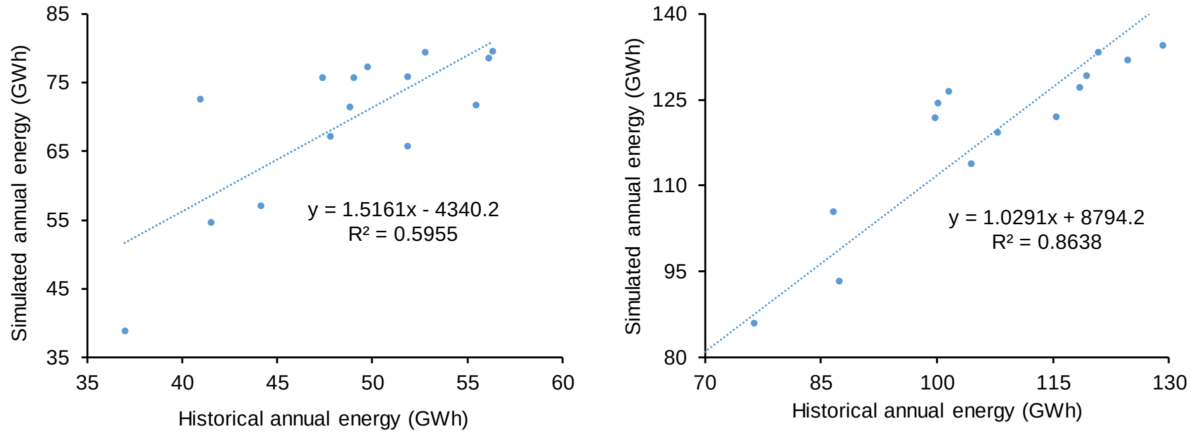


Figure A1. The results of the correlation coefficients (r^2) for Milford Dam (a) West Enfield Dam (b).

Section A4. Behaviour test of age-structured fish population model

The potentially historical minimum and maximum alewife spawner populations entering rivers in Maine (in millions) were calculated according to the minimum and maximum alewife historical landing data (in million pounds), average alewife spawner weight (in pounds), and alewife harvest rate which was assumed in the range of 10-70% (Barber et al., 2018; MaineDMR, 2016) (Equation A2 and A3):

$$P_{min} = \frac{L_{min}}{W} \times \frac{1}{0.7} \quad \text{(Equation A2)}$$

$$P_{max} = \frac{L_{max}}{W} \times \frac{1}{0.1} \quad \text{(Equation A3)}$$

where P_{min} and P_{max} are the extrapolated historical minimum and maximum alewife spawner population entering rivers in Maine, respectively, million; W is the average weight of alewife spawners, 0.4 pounds (Barber et al., 2018); L_{min} and L_{max} are minimum and maximum alewife landings in Maine, million popunds. According to data collected by the Department of Marine Resources in the period of 1950 to 2016 (MaineDMR, 2018), L_{min} and L_{max} are 0.15 and 4.6 million pounds, respectively. Thus, the historical minimum and maximum alewife spawner population entering rivers in Maine are 0.5 and 115 million, respectively.

For the simulated spawner population entering the Penobscot River, its minimum and maximum values were determined by changing fish upstream and downstream passage rates (0-100%), as well as fishing mortalities (0-70%). The fish model was initialized with 1 million juveniles entering the ocean. The results shown that the simulated minimum and maximum alewife spawner population entering the Penobscot River are 0 and 25.6 million, respectively, that are within the range of historical alewives entering rivers in Maine. Therefore, we think this fish population model could reflect the current alewife population status and passes the behaviour test.

Section A5. Equations in the energy model

Dam-*i* impoundment storage = INTEG (river flow-*i* − fishway-*i* baseflow − spillway release-*i* − actual turbine-*i* release,1.76696e+007)
river flow-*i* = GET XLS DATA ('River discharge from USGS.xlsx', 'Daily discharge' , 'A' , 'S3')

fishway-*i* baseflow = IF THEN ELSE (install fishway-*i* =1:AND:upstream migration period=1, fishway-*i* attraction flow, 0)
upstream migration period = PULSE TRAIN(82, 87, 365, 54750)
fishway-*i* attraction flow = IF THEN ELSE(attraction flow based on river flow-*i* >=attraction flow based on turbine-*i* release, attraction flow based on river flow-*i* ,attraction flow based on turbine-*i* release)
attraction flow based on turbine-*i* release = fishway attraction flow ratio*Maximum release capacity-*i*
attraction flow based on river flow-*i* = fishway attraction flow ratio*annual mean river flow-*i*
fishway attration flow ratio = 0.05
flows to turbines-*i* = river flow-*i* − fishway-*i* baseflow
actual turbine-*i* release = IF THEN ELSE(remove dam-*i* =1, 0 , IF THEN ELSE(turbine-*i* shutdown=1:AND:turbine shutdown period=1, 0 , turbine-*i* release))
shutdown period for adults = PULSE TRAIN(141, 10, 365, 54750)
shutdown period for juvenile = PULSE TRAIN(231, 10 , 365 , 54750)
turbine shutdown period = shutdown period for adults + shutdown period for juvenile
turbine-*i* release = IF THEN ELSE(flows to turbines-*i* ≥ Maximum release capacity-*i* , Maximum release capacity-*i*, IF THEN ELSE (flows to turbines-*i* < Minimum release capacity-*i* , 0 , flows to turbines-*i*))
spillway release-*i* = flows to turbines-*i* − actual turbine-*i* release
dam-*i* actual hydropower = overall plant efficiency1 × net water head1 × actual turbine1 release × 1000 × 9.81/1e+006 × unit factor
unit factor = 1/24/3600
dam-*i* actual energy = dam-*i* actual hydropower × turbine-*i* daily operation period
turbine1 daily operation period = 24

Section A6. Equations in the age-structured fish population model

Equations in the age-structure Alewife model

Juveniles in Dmigration = INTEG(juveniles leave river - juvenile loss - juveniles enter ocean , 0)
juveniles leave river = DELAY FIXED (total alewife age0 recruits ,DT of juveniles in FW , 0)
total alewife age0 recruits = alewife age0 recruits HA1 + alewife age0 recruits HA2 + alewife age0 recruits HA3 + alewife age0 recruits HA4 + alewife age0 recruits HA5 + alewife age0 recruits HA6
juvenile loss = juveniles loss in Dmigration
juveniles loss in Dmigration = juveniles loss at dam1 + juveniles loss at dam2 + juveniles loss at dam3 + juveniles loss at dam4 + juveniles loss at dam5 **(see alewife juvenile downstream migration model)**
juveniles enter ocean = juveniles arrive estuary
juveniles arrive estuary = juveniles at HA1 + juveniles leave dam1 **(see alewife juvenile downstream migration model)**
Juvenile to age3 alewife in ocean = INTEG (juveniles enter ocean - age3 alewife loss in ocean - immature age3 alewife - age3 spawners return to river, 1e+006)
age3 alewife loss in ocean = delayed 3yr alewife loss + stocking 3yr alewife loss
age3 mature probability = 0.35
age3 spawn loss = age3 spawners arrive HAs * spawning mortality
age3 spawner ratio during Dmigration = age3 spawner ratio + age3 spawner ratio dam1 + age3 spawner ratio dam2 + age3 spawner ratio dam3 + age3 spawner ratio dam4 + age3 spawner ratio dam5
Age3 spawners about to spawn = INTEG(age3 spawners arrive HAs - age3 spawn loss - surviving age3 spawner , 0)
age3 spawners arrive HAs = delayed age3 spawners to HAs

age3 spawners Dmigration loss = spawners loss in Dmigration * age3 spawner ratio during Dmigration
age3 spawners enter ocean = spawners arrive estuary * age3 spawner ratio during Dmigration
Age3 spawners in Dmigration = INTEG(surviving age3 spawner - age3 spawners enter ocean - age3 spawners Dmigration loss , 0)
Age3 spawners in Umigration = INTEG(age3 spawners return to river - age3 spawners arrive HAs , 0)
age3 spawners return to river = delayed 3yr alewife to spawn + stocking 3yr alewife to spawn
age4 alewife loss in ocean = total 4yr alewife * (1 - EXP (-0.92 * ocean mortality))
age4 mature probability = 0.51
age4 spawn loss = age4 spawners arrive HAs * spawning mortality
age4 spawner ratio during Dmigration = age4 spawner ratio + age4 spawner ratio dam5 + age4 spawner ratio dam4 + age4 spawner ratio dam3 + age4 spawner ratio dam2 + age4 spawner ratio dam1
Age4 spawners about to spawn = INTEG(age4 spawners arrive HAs - age4 spawn loss - survivingage4 spawners , 0)
age4 spawners arrive HAs = delayed age4 spawners to HAs
age4 spawners Dmigration loss = spawners loss in Dmigration * age4 spawner ratio during Dmigration
age4 spawners enter ocean = spawners arrive estuary * age4 spawner ratio during Dmigration
Age 4 spawners in Dmigration = INTEG(survivingage4 spawners - age4 spawners enter ocean - age4 spawners Dmigration loss , 0)
Age4 spawners in Umigration = INTEG(age4 spawners return to river - age4 spawners arrive HAs , 0)
age4 spawners return to river = total 4yr alewife * EXP (-0.92 * ocean mortality) * age4 mature probability
age5 alewife loss in ocean = total 5yr alewife * (1 - EXP (-0.92 * ocean mortality))
age5 mature probability = 0.96
age5 spawn loss = age5 spawners arrive HAs * spawning mortality
age5 spawner ratio during Dmigration = age5 spawner ratio + age5 spawner ratio dam5 + age5 spawner ratio dam4 + age5 spawner ratio dam3 + age5 spawner ratio dam2 + age5 spawner ratio dam1
Age5 spawners about to spawn = INTEG(age5 spawners arrive HAs - age5 spawn loss - survivingage5 spawners , 0)
age5 spawners arrive HAs = delayed age5 spawners to HAs
age5 spawners Dmigration loss = spawners loss in Dmigration * age5 spawner ratio during Dmigration
age5 spawners enter ocean = spawners arrive estuary * age5 spawner ratio during Dmigration
Age5 spawners in Dmigration = INTEG(survivingage5 spawners - age5 spawners enter ocean - age5 spawners Dmigration loss , 0)
Age5 spawners in Umigration = INTEG(age5 spawners return to river - age5 spawners arrive HAs , 0)
age5 spawners return to river = total 5yr alewife * EXP (-0.92 * ocean mortality) * age5 mature probability
age6 alewife loss in ocean = total 6yr alewife * (1 - EXP (-0.92 * ocean mortality))
age6 mature probability = 0.96
age6 spawn loss = age6 spawners arrive HAs * spawning mortality
age6 spawner ratio during Dmigration = age6 spawner ratio + age6 spawner ratio dam1 + age6 spawner ratio dam2 + age6 spawner ratio dam3 + age6 spawner ratio dam4 + age6 spawner ratio dam5
Age6 spawners about to spawn = INTEG(age6 spawners arrive HAs - age6 spawn loss - surviving age6 spawners , 0)
age6 spawners arrive HAs = delayed age6 spawners to HAs
age6 spawners Dmigration loss = age6 spawner ratio during Dmigration * spawners loss in Dmigration
age6 spawners enter ocean = spawners arrive estuary * age6 spawner ratio during Dmigration
Age6 spawners in Dmigration = INTEG(surviving age6 spawners - age6 spawners enter ocean - age6 spawners Dmigration loss , 0)
Age6 spawners in Umigration = INTEG(age6 spawners return to river - age6 spawners arrive HAs , 0)
age6 spawners return to river = total 6yr alewife * EXP (-0.92 * ocean mortality) * age6 mature probability
alewife 4yr from immature = DELAY FIXED (immature age3 alewife ,DT of alewife maturation , 0)
alewife 4yr from spawner = DELAY FIXED (returning 3yr spawners ,DT of spawner growth , 0)
Alewife 4yr olds in ocean = INTEG(immature age3 alewife + returning 3yr spawners - age4 spawners

return to river - age4 alewife loss in ocean - immature age4 alewife , 0)
alewife 5yr from immature = DELAY FIXED (immature age4 alewife ,DT of alewife maturation , 0)
alewife 5yr from spawner = DELAY FIXED (returning 4yr spawners ,DT of spawner growth , 0)
Alewife 5yr olds in ocean = INTEG(immature age4 alewife + returning 4yr spawners - age5 alewife loss in ocean - age5 spawners return to river - immature age5 alewife , 0)
alewife 6yr from immature = DELAY FIXED (immature age5 alewife ,DT of alewife maturation , 0)
alewife 6yr from spawner = DELAY FIXED (returning 5yr spawners ,DT of spawner growth , 0)
Alewife 6yr olds in ocean = INTEG(returning 5yr spawners + immature age5 alewife - age6 alewife loss in ocean - age6 spawners return to river - immature age6 alewife , 0)
delayed 3yr alewife = DELAY FIXED (juveniles enter ocean ,DT of juvenile to age3 , 0)
delayed 3yr alewife loss = delayed 3yr alewife * (1 - EXP (- ocean mortality * 945 / 365))
delayed 3yr alewife to spawn = delayed 3yr alewife * age3 mature probability * EXP (- ocean mortality * 945 / 365)
delayed age3 spawners to HAs = DELAY FIXED (age3 spawners return to river * (1 - fishing mortality) ,DT of Umigration , 0)
delayed age4 spawners to HAs = DELAY FIXED (age4 spawners return to river * (1 - fishing mortality) ,DT of Umigration , 0)
delayed age5 spawners to HAs = DELAY FIXED (age5 spawners return to river * (1 - fishing mortality) ,DT of Umigration , 0)
delayed age6 spawners to HAs = DELAY FIXED (age6 spawners return to river * (1 - fishing mortality) ,DT of Umigration , 0)
delayed immature 3yr alewife = delayed 3yr alewife * (1 - age3 mature probability) * EXP (- ocean mortality * 945 / 365)
DT of alewife maturation = 365
DT of juvenile to age3 = 975
DT of juveniles in FW = 90
DT of spawner growth = 335
DT of Umigration = 20
fishing mortality = 0.4
immature age3 alewife = delayed immature 3yr alewife + stocking immature 3yr alewife
immature age4 alewife = total 4yr alewife * EXP (-0.92 * ocean mortality) * (1 - age4 mature probability)
immature age5 alewife = total 5yr alewife * EXP (-0.92 * ocean mortality) * (1 - age5 mature probability)
immature age6 alewife = total 6yr alewife * EXP (-0.92 * ocean mortality) * (1 - age6 mature probability)
ocean mortality = 0.648
returning 3yr spawners = age3 spawners enter ocean
returning 4yr spawners = age4 spawners enter ocean
returning 5yr spawners = age5 spawners enter ocean
spawners arrive estuary = survived spawners at HA1 + spawners leave dam1
spawners in the ocean = age3 spawners return to river + age4 spawners return to river + age5 spawners return to river + age6 spawners return to river
spawners loss in Dmigration = spawners loss at dam1 + spawners loss at dam2 + spawners loss at dam3 + spawners loss at dam4 + spawners loss at dam5
spawning mortality = 0.45
stocking 3yr alewife loss = IF THEN ELSE (Time = 120, stocking juveniles * (1 - EXP (- ocean mortality * 975 / 365)) , 0)
stocking 3yr alewife to spawn = IF THEN ELSE (Time = 120, stocking juveniles * EXP (- ocean mortality * 975 / 365) * age3 mature probability , 0)
stocking immature 3yr alewife = IF THEN ELSE (Time = 120, stocking juveniles * EXP (- ocean mortality * 975 / 365) * (1 - age3 mature probability) , 0)
stocking juveniles = 1e+006
surviving age3 spawner = age3 spawners arrive HAs * (1 - spawning mortality)

surviving age6 spawners = age6 spawners arrive HAs * (1 - spawning mortality)
survivingage4 spawners = age4 spawners arrive HAs * (1 - spawning mortality)
survivingage5 spawners = age5 spawners arrive HAs * (1 - spawning mortality)
total 4yr alewife = alewife 4yr from immature + alewife 4yr from spawner
total 5yr alewife = alewife 5yr from immature + alewife 5yr from spawner
total 6yr alewife = alewife 6yr from spawner + alewife 6yr from immature
total spawners = surviving age3 spawner + survivingage4 spawners + survivingage5 spawners + surviving age6 spawners
spawners in the ocean = age3 spawners return to river + age4 spawners return to river + age5 spawners return to river + age6 spawners return to river

Alewife upstream migration model
age3 spawner ratio = IF THEN ELSE (total spawners = 0, 0, surviving age3 spawner / total spawners)
age3 spawner ratio dam1 = DELAY FIXED (age3 spawner ratio ,5 * DT of dam alewife , 0)
age3 spawner ratio dam2 = DELAY FIXED (age3 spawner ratio ,4 * DT of dam alewife , 0)
age3 spawner ratio dam3 = DELAY FIXED (age3 spawner ratio ,3 * DT of dam alewife , 0)
age3 spawner ratio dam4 = DELAY FIXED (age3 spawner ratio ,2 * DT of dam alewife , 0)
age3 spawner ratio dam5 = DELAY FIXED (age3 spawner ratio ,DT of dam alewife , 0)
age3 spawner ratio during Dmigration = age3 spawner ratio + age3 spawner ratio dam1 + age3 spawner ratio dam2 + age3 spawner ratio dam3 + age3 spawner ratio dam4 + age3 spawner ratio dam5
age3 spawner weight = 144
age3 spawners arrive HAs = delayed age3 spawners to HAs
age4 spawner ratio = IF THEN ELSE (total spawners = 0, 0, survivingage4 spawners / total spawners)
age4 spawner ratio dam1 = DELAY FIXED (age4 spawner ratio ,5 * DT of dam alewife , 0)
age4 spawner ratio dam2 = DELAY FIXED (age4 spawner ratio ,4 * DT of dam alewife , 0)
age4 spawner ratio dam3 = DELAY FIXED (age4 spawner ratio ,3 * DT of dam alewife , 0)
age4 spawner ratio dam4 = DELAY FIXED (age4 spawner ratio ,2 * DT of dam alewife , 0)
age4 spawner ratio dam5 = DELAY FIXED (age4 spawner ratio ,DT of dam alewife , 0)
age4 spawner ratio during Dmigration = age4 spawner ratio + age4 spawner ratio dam5 + age4 spawner ratio dam4 + age4 spawner ratio dam3 + age4 spawner ratio dam2 + age4 spawner ratio dam1
age4 spawner weight = 186
age4 spawners arrive HAs = delayed age4 spawners to HAs
age5 spawner ratio = IF THEN ELSE (total spawners = 0, 0, survivingage5 spawners / total spawners)
age5 spawner ratio dam1 = DELAY FIXED (age5 spawner ratio ,5 * DT of dam alewife , 0)
age5 spawner ratio dam2 = DELAY FIXED (age5 spawner ratio ,4 * DT of dam alewife , 0)
age5 spawner ratio dam3 = DELAY FIXED (age5 spawner ratio ,3 * DT of dam alewife , 0)
age5 spawner ratio dam4 = DELAY FIXED (age5 spawner ratio ,2 * DT of dam alewife , 0)
age5 spawner ratio dam5 = DELAY FIXED (age5 spawner ratio ,DT of dam alewife , 0)
age5 spawner ratio during Dmigration = age5 spawner ratio + age5 spawner ratio dam5 + age5 spawner ratio dam4 + age5 spawner ratio dam3 + age5 spawner ratio dam2 + age5 spawner ratio dam1
age5 spawner weight = 209
age5 spawners arrive HAs = delayed age5 spawners to HAs
age6 spawner ratio = IF THEN ELSE (total spawners = 0, 0, surviving age6 spawners / total spawners)
age6 spawner ratio dam1 = DELAY FIXED (age6 spawner ratio ,5 * DT of dam alewife , 0)
age6 spawner ratio dam2 = DELAY FIXED (age6 spawner ratio ,4 * DT of dam alewife , 0)
age6 spawner ratio dam3 = DELAY FIXED (age6 spawner ratio ,3 * DT of dam alewife , 0)
age6 spawner ratio dam4 = DELAY FIXED (age6 spawner ratio ,2 * DT of dam alewife , 0)
age6 spawner ratio dam5 = DELAY FIXED (age6 spawner ratio ,DT of dam alewife , 0)
age6 spawner ratio during Dmigration = age6 spawner ratio + age6 spawner ratio dam1 + age6 spawner ratio dam2 + age6 spawner ratio dam3 + age6 spawner ratio dam4 + age6 spawner ratio dam5
age6 spawner weight = 244

alewife age0 recruits HA1 = alewife alpha * alewife eggs HA1 / (1 + (alewife alpha * alewife eggs HA1 / (total potential alewife habitat * ratio of HA1 * alewife recruits per HU)))
alewife age0 recruits HA2 = IF THEN ELSE (ratio of HA2 = 0, 0, alewife alpha * alewife eggs HA2 / (1 + (alewife alpha * alewife eggs HA2 / (ratio of HA2 * total potential alewife habitat * alewife recruits per HU))))
alewife age0 recruits HA3 = IF THEN ELSE (ratio of HA3 = 0, 0, alewife alpha * alewife eggs HA3 / (1 + (alewife alpha * alewife eggs HA3 / (ratio of HA3 * total potential alewife habitat * alewife recruits per HU))))
alewife age0 recruits HA4 = IF THEN ELSE (ratio of HA4 = 0, 0, alewife alpha * alewife eggs HA4 / (1 + (alewife alpha * alewife eggs HA4 / (ratio of HA4 * total potential alewife habitat * alewife recruits per HU))))
alewife age0 recruits HA5 = IF THEN ELSE (ratio of HA5 = 0, 0, alewife alpha * alewife eggs HA5 / (1 + (alewife alpha * alewife eggs HA5 / (ratio of HA5 * total potential alewife habitat * alewife recruits per HU))))
alewife age0 recruits HA6 = IF THEN ELSE (ratio of HA6 = 0, 0, alewife alpha * alewife eggs HA6 / (1 + (alewife alpha * alewife eggs HA6 / (ratio of HA6 * total potential alewife habitat * alewife recruits per HU))))
alewife alpha = 0.0015
alewife dam1 pass rate = IF THEN ELSE (remove dam1 = 1, 1, IF THEN ELSE (install fishway1 = 1, fishway pass rate , 0))
alewife dam2 pass rate = IF THEN ELSE (remove dam2 = 1, 1, IF THEN ELSE (install fishway2 = 1, fishway pass rate , 0))
alewife dam3 pass rate = IF THEN ELSE (remove dam3 = 1, 1, IF THEN ELSE (install fishway3 = 1, fishway pass rate , 0))
alewife dam4 pass rate = IF THEN ELSE (remove dam4 = 1, 1, IF THEN ELSE (install fishway4 = 1, fishway pass rate , 0))
alewife dam5 pass rate = IF THEN ELSE (remove dam5 = 1, 1, IF THEN ELSE (install fishway5 = 1, fishway pass rate , 0))
alewife eggs HA1 = alewife sex ratio * alewife spawner HA1 * probability of spawning * (age3 spawner ratio * (fecundity slope * age3 spawner weight - fecundity intercept) + age4 spawner ratio * (fecundity slope * age4 spawner weight - fecundity intercept) + age5 spawner ratio * (fecundity slope * age5 spawner weight - fecundity intercept) + age6 spawner ratio * (fecundity slope * age6 spawner weight - fecundity intercept))
alewife eggs HA2 = alewife sex ratio * alewife spawner HA2 * probability of spawning * (age3 spawner ratio * (fecundity slope * age3 spawner weight - fecundity intercept) + age4 spawner ratio * (fecundity slope * age4 spawner weight - fecundity intercept) + age5 spawner ratio * (fecundity slope * age5 spawner weight - fecundity intercept) + age6 spawner ratio * (fecundity slope * age6 spawner weight - fecundity intercept))
alewife eggs HA3 = alewife sex ratio * alewife spawner HA3 * probability of spawning * (age3 spawner ratio * (fecundity slope * age3 spawner weight - fecundity intercept) + age4 spawner ratio * (fecundity slope * age4 spawner weight - fecundity intercept) + age5 spawner ratio * (fecundity slope * age5 spawner weight - fecundity intercept) + age6 spawner ratio * (fecundity slope * age6 spawner weight - fecundity intercept))
alewife eggs HA4 = alewife sex ratio * alewife spawner HA4 * probability of spawning * (age3 spawner ratio * (fecundity slope * age3 spawner weight - fecundity intercept) + age4 spawner ratio * (fecundity slope * age4 spawner weight - fecundity intercept) + age5 spawner ratio * (fecundity slope * age5 spawner weight - fecundity intercept) + age6 spawner ratio * (fecundity slope * age6 spawner weight - fecundity intercept))
alewife eggs HA5 = alewife sex ratio * alewife spawner HA5 * probability of spawning * (age3 spawner ratio * (fecundity slope * age3 spawner weight - fecundity intercept) + age4 spawner ratio * (fecundity slope * age4 spawner weight - fecundity intercept) + age5 spawner ratio * (fecundity slope * age5 spawner

weight - fecundity intercept) + age6 spawner ratio * (fecundity slope * age6 spawner weight - fecundity intercept))
alewife eggs HA6 = alewife sex ratio * alewife spawner HA6 * probability of spawning * (age3 spawner ratio * (fecundity slope * age3 spawner weight - fecundity intercept) + age4 spawner ratio * (fecundity slope * age4 spawner weight - fecundity intercept) + age5 spawner ratio * (fecundity slope * age5 spawner weight - fecundity intercept) + age6 spawner ratio * (fecundity slope * age6 spawner weight - fecundity intercept))
alewife recruits per HU = 3283
alewife sex ratio = 0.5
alewife spawner HA1 = ratio of HA1 * total spawners + (1 - ratio of HA1) * total spawners * (1 - alewife dam1 pass rate)
alewife spawner HA2 = IF THEN ELSE ((1 - ratio of HA1) * alewife dam1 pass rate <= 0, 0, IF THEN ELSE ((1 - ratio of HA1) * alewife dam1 pass rate <= ratio of HA2 , (1 - ratio of HA1) * alewife dam1 pass rate * total spawners , ratio of HA2 * total spawners + ((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * total spawners * (1 - alewife dam2 pass rate)))
alewife spawner HA3 = IF THEN ELSE (((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate <= 0, 0, IF THEN ELSE (((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate <= ratio of HA3 , ((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate * total spawners , ratio of HA3 * total spawners + (((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * total spawners * (1 - alewife dam3 pass rate)))
alewife spawner HA4 = IF THEN ELSE ((((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate <= 0, 0, IF THEN ELSE ((((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate <= ratio of HA4 , (((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate * total spawners , ratio of HA4* total spawners + ((((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate - ratio of HA4) * total spawners * (1 - alewife dam4 pass rate)))
alewife spawner HA5 = IF THEN ELSE (((((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate - ratio of HA4) * alewife dam4 pass rate <= 0, 0, IF THEN ELSE (((((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate - ratio of HA4) * alewife dam4 pass rate <= ratio of HA5 , ((((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate - ratio of HA4) * alewife dam4 pass rate * total spawners , ratio of HA5 * total spawners + (((((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate - ratio of HA4) * alewife dam4 pass rate - ratio of HA5) * total spawners * (1 - alewife dam5 pass rate)))
alewife spawner HA6 = IF THEN ELSE ((((((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate - ratio of HA4) * alewife dam4 pass rate - ratio of HA5) * alewife dam5 pass rate <= 0, 0, (((((1 - ratio of HA1) * alewife dam1 pass rate - ratio of HA2) * alewife dam2 pass rate - ratio of HA3) * alewife dam3 pass rate - ratio of HA4) * alewife dam4 pass rate - ratio of HA5) * alewife dam5 pass rate * total spawners)
DT of dam alewife = 2
DT of juveniles in FW = 90
fecundity intercept = 50916
fecundity slope = 871.72
fishway pass rate = 1
fraction of HA1 juveniles = DELAY FIXED (fraction of recruits at HA1 ,DT of juveniles in FW , 0)
fraction of HA2 juveniles = DELAY FIXED (fraction of recruits at HA2 ,DT of juveniles in FW , 0)
fraction of HA3 juveniles = DELAY FIXED (fraction of recruits at HA3 ,DT of juveniles in FW , 0)
fraction of HA4 juveniles = DELAY FIXED (fraction of recruits at HA4 ,DT of juveniles in FW , 0)

fraction of HA5 juveniles = DELAY FIXED (fraction of recruits at HA5 ,DT of juveniles in FW , 0)
fraction of HA6 juveniles = DELAY FIXED (fraction of recruits at HA6 ,DT of juveniles in FW , 0)
fraction of recruits at HA1 = IF THEN ELSE (total alewife age0 recruits = 0, 0, alewife age0 recruits HA1 / total alewife age0 recruits)
fraction of recruits at HA2 = IF THEN ELSE (total alewife age0 recruits = 0, 0, alewife age0 recruits HA2 / total alewife age0 recruits)
fraction of recruits at HA3 = IF THEN ELSE (total alewife age0 recruits = 0, 0, alewife age0 recruits HA3 / total alewife age0 recruits)
fraction of recruits at HA4 = IF THEN ELSE (total alewife age0 recruits = 0, 0, alewife age0 recruits HA4 / total alewife age0 recruits)
fraction of recruits at HA5 = IF THEN ELSE (total alewife age0 recruits = 0, 0, alewife age0 recruits HA5 / total alewife age0 recruits)
fraction of recruits at HA6 = IF THEN ELSE (total alewife age0 recruits = 0, 0, alewife age0 recruits HA6 / total alewife age0 recruits)
probability of spawning = 0.95
ratio of HA1 = 0.1
ratio of HA2 = 0.03
ratio of HA3 = 0
ratio of HA4 = 0.36
ratio of HA5 = 0.32
ratio of HA6 = 0.19
spawning mortality = 0.45
surviving age3 spawner = age3 spawners arrive HAs * (1 - spawning mortality)
surviving age6 spawners = age6 spawners arrive HAs * (1 - spawning mortality)
survivingage4 spawners = age4 spawners arrive HAs * (1 - spawning mortality)
survivingage5 spawners = age5 spawners arrive HAs * (1 - spawning mortality)
total alewife age0 recruits = alewife age0 recruits HA1 + alewife age0 recruits HA2 + alewife age0 recruits HA3 + alewife age0 recruits HA4 + alewife age0 recruits HA5 + alewife age0 recruits HA6
total potential alewife habitat = 81393
total spawners = surviving age3 spawner + survivingage4 spawners + survivingage5 spawners + surviving age6 spawners
DT of juvenile to age3 = 975
DT of Umigration = 20
ocean mortality = 0.648

Alewife spawner downstream migration model

Alewife spawners in reservoir 1 = INTEG(survived spawners at HA2 + spawners leave dam2 - spawners leave dam1 - spawners loss at dam1 , 0)
Alewife spawners in reservoir 2 = INTEG(survived spawners at HA3 + spawners leave dam3 - spawners leave dam2 - spawners loss at dam2 , 0)
Alewife spawners in reservoir 3 = INTEG(survived spawners at HA4 + spawners leave dam4 - spawners leave dam3 - spawners loss at dam3 , 0)
Alewife spawners in reservoir 4 = INTEG(survived spawners at HA5 + spawners leave dam5 - spawners leave dam4 - spawners loss at dam4 , 0)
Alewife spawners in reservoir 5 = INTEG(Survived spawners at HA6 - spawners leave dam5 - spawners loss at dam5 , 0)
total alewife spawners at reservoir1 = DELAY FIXED ((survived spawners at HA2 + spawners leave dam2) ,DT of dam alewife , 0)
total alewife spawners at reservoir2 = DELAY FIXED ((survived spawners at HA3 + spawners leave dam3) ,DT of dam alewife , 0)
total alewife spawners at reservoir3 = DELAY FIXED ((spawners leave dam4 + survived spawners at

HA4) ,DT of dam alewife , 0)
total alewife spawners at reservoir4 = DELAY FIXED ((spawners leave dam5 + survived spawners at HA5) ,DT of dam alewife , 0)
total alewife spawners at reservoir5 = DELAY FIXED (Survived spawners at HA6 ,DT of dam alewife , 0)
spawners arrive estuary = survived spawners at HA1 + spawners leave dam1
spawners leave dam1 = total alewife spawners at reservoir1 - spawners loss at dam1
spawners leave dam2 = total alewife spawners at reservoir2 - spawners loss at dam2
spawners leave dam3 = total alewife spawners at reservoir3 - spawners loss at dam3
spawners leave dam4 = total alewife spawners at reservoir4 - spawners loss at dam4
spawners leave dam5 = total alewife spawners at reservoir5 - spawners loss at dam5
spawners loss at dam1 = total alewife spawners at reservoir1 * fraction of spawners enter turbine1 * spawners turbine mortality
spawners loss at dam2 = total alewife spawners at reservoir2 * fraction of spawners enter turbine2 * spawners turbine mortality
spawners loss at dam3 = total alewife spawners at reservoir3 * fraction of spawners enter turbine3 * spawners turbine mortality
spawners loss at dam4 = total alewife spawners at reservoir4 * fraction of spawners enter turbine4 * spawners turbine mortality
spawners loss at dam5 = total alewife spawners at reservoir5 * fraction of spawners enter turbine5 * spawners turbine mortality
spawners loss in Dmigration = spawners loss at dam1 + spawners loss at dam2 + spawners loss at dam3 + spawners loss at dam4 + spawners loss at dam5
spawners turbine mortality = 0.3
DT of dam alewife = 2
fraction of spawners enter turbine1 = actual turbine1 release / river flow1
fraction of spawners enter turbine2 = actual turbine2 release / river flow2
fraction of spawners enter turbine3 = actual turbine3 release / river flow3
fraction of spawners enter turbine4 = actual turbine4 release / river flow4
fraction of spawners enter turbine5 = actual turbine5 release / river flow5
survived spawners at HA1 = DELAY FIXED (alewife spawner HA1 ,5 * DT of dam alewife , 0)
survived spawners at HA2 = DELAY FIXED (alewife spawner HA2 ,4 * DT of dam alewife , 0)
survived spawners at HA3 = DELAY FIXED (alewife spawner HA3 ,3 * DT of dam alewife , 0)
survived spawners at HA4 = DELAY FIXED (alewife spawner HA4 ,2 * DT of dam alewife , 0)
survived spawners at HA5 = DELAY FIXED (alewife spawner HA5 ,DT of dam alewife , 0)
Survived spawners at HA6 = alewife spawner HA6

Alewife juvenile downstream migration model

Alewife juveniles in reservoir 1 = INTEG(juveniles at HA2 + juveniles leave dam2 - juveniles leave dam1 - juveniles loss at dam1 , 0)
Alewife juveniles in reservoir 2 = INTEG(juveniles at HA3 + juveniles leave dam3 - juveniles leave dam2 - juveniles loss at dam2 , 0)
Alewife juveniles in reservoir 3 = INTEG(juveniles at HA4 + juveniles leave dam4 - juveniles leave dam3 - juveniles loss at dam3 , 0)
Alewife juveniles in reservoir 4 = INTEG(juveniles at HA5 + juveniles leave dam5 - juveniles leave dam4 - juveniles loss at dam4 , 0)
Alewife juveniles in reservoir 5 = INTEG(juveniles at HA6 - juveniles leave dam5 - juveniles loss at dam5 , 0)
juveniles at HA1 = DELAY FIXED (fraction of HA1 juveniles * juveniles leave river ,5 * DT of dam alewife , 0)
juveniles at HA2 = DELAY FIXED (fraction of HA2 juveniles * juveniles leave river ,4 * DT of dam alewife , 0)

juveniles at HA3 = DELAY FIXED (fraction of HA3 juveniles * juveniles leave river ,3 * DT of dam alewife , 0)
juveniles at HA4 = DELAY FIXED (fraction of HA4 juveniles * juveniles leave river ,2 * DT of dam alewife , 0)
juveniles at HA5 = DELAY FIXED (fraction of HA5 juveniles * juveniles leave river ,DT of dam alewife , 0)
juveniles at HA6 = fraction of HA6 juveniles * juveniles leave river
fraction of HA1 juveniles = DELAY FIXED (fraction of recruits at HA1 ,DT of juveniles in FW , 0)
fraction of HA2 juveniles = DELAY FIXED (fraction of recruits at HA2 ,DT of juveniles in FW , 0)
fraction of HA3 juveniles = DELAY FIXED (fraction of recruits at HA3 ,DT of juveniles in FW , 0)
fraction of HA4 juveniles = DELAY FIXED (fraction of recruits at HA4 ,DT of juveniles in FW , 0)
fraction of HA5 juveniles = DELAY FIXED (fraction of recruits at HA5 ,DT of juveniles in FW , 0)
fraction of HA6 juveniles = DELAY FIXED (fraction of recruits at HA6 ,DT of juveniles in FW , 0)
DT of dam alewife = 2
juveniles leave river = DELAY FIXED (total alewife age0 recruits ,DT of juveniles in FW , 0)
juveniles leave dam1 = total alewife juveniles at reservoir1 - juveniles loss at dam1
juveniles leave dam2 = total alewife juveniles at reservoir2 - juveniles loss at dam2
juveniles leave dam3 = total alewife juveniles at reservoir3 - juveniles loss at dam3
juveniles leave dam4 = total alewife juveniles at reservoir4 - juveniles loss at dam4
juveniles leave dam5 = total juveniles at reservoir5 - juveniles loss at dam5
juveniles loss at dam1 = total alewife juveniles at reservoir1 * fraction of juveniles enter turbine1 * juveniles turbine mortality
juveniles loss at dam2 = total alewife juveniles at reservoir2 * fraction of juveniles enter turbine2 * juveniles turbine mortality
juveniles loss at dam3 = total alewife juveniles at reservoir3 * fraction of juveniles enter turbine3 * juveniles turbine mortality
juveniles loss at dam4 = total alewife juveniles at reservoir4 * juveniles turbine mortality * fraction of juveniles enter turbine4
juveniles loss at dam5 = total juveniles at reservoir5 * fraction of juveniles enter turbine5 * juveniles turbine mortality
juveniles loss in Dmigration = juveniles loss at dam1 + juveniles loss at dam2 + juveniles loss at dam3 + juveniles loss at dam4 + juveniles loss at dam5
juveniles turbine mortality = 0.3
fraction of juveniles enter turbine1 = actual turbine1 release / river flow1
fraction of juveniles enter turbine2 = actual turbine2 release / river flow2
fraction of juveniles enter turbine3 = actual turbine3 release / river flow3
fraction of juveniles enter turbine4 = actual turbine4 release / river flow4
fraction of juveniles enter turbine5 = actual turbine5 release / river flow5
juveniles arrive estuary = juveniles at HA1 + juveniles leave dam1
total alewife juveniles at reservoir1 = DELAY FIXED ((juveniles at HA2 + juveniles leave dam2) ,DT of dam alewife , 0)
total alewife juveniles at reservoir2 = DELAY FIXED ((juveniles at HA3 + juveniles leave dam3) ,DT of dam alewife , 0)
total alewife juveniles at reservoir3 = DELAY FIXED ((juveniles leave dam4 + juveniles at HA4) ,DT of dam alewife , 0)
total alewife juveniles at reservoir4 = DELAY FIXED ((juveniles leave dam5 + juveniles at HA5) ,DT of dam alewife , 0)
total juveniles at reservoir5 = DELAY FIXED (juveniles at HA6 ,DT of dam alewife , 0)

Section A7. Tradeoffs between energy and alewife juveniles

Juvenile abundance is a commonly applied indicator to assess the effectiveness of restoration programs (McHenry and Pess, 2008). Figure A2 illustrates the tradeoffs between annual hydropower generation and the stabilized juvenile abundance under the eight dam management scenarios. Generally, the restoration effects of different basin-scale dam management scenarios on spawners and juveniles are similar except the F and PR-PF scenarios. This similarity in population changes between juvenile and spanwer are reasonable because the abundance of juveniles in the freshwater grounds is directly determined by spawners abundance according to the B-H recruitment curve (Equation 3). Under the F and PR-PF scenarios, juvenile abundances were increased by 107-119% and 331-361%, respectively, compared to the NR scenario. Spawner abundance only increased 45-48% and 225-236% correspondingly. This difference could be caused by the fact that juvenile abundance we report here are juvenile in spawner HAs, they have not subjected to cumulative dam kills during downstream migration. From juveniles' perspective, the PR-PF-S scenario is also the best dam management option in balancing migratory fish restoration and energy loss, which restored 58-65% of juvenile abundance while preserving around 65% hydropower generation compared to the R scenario.

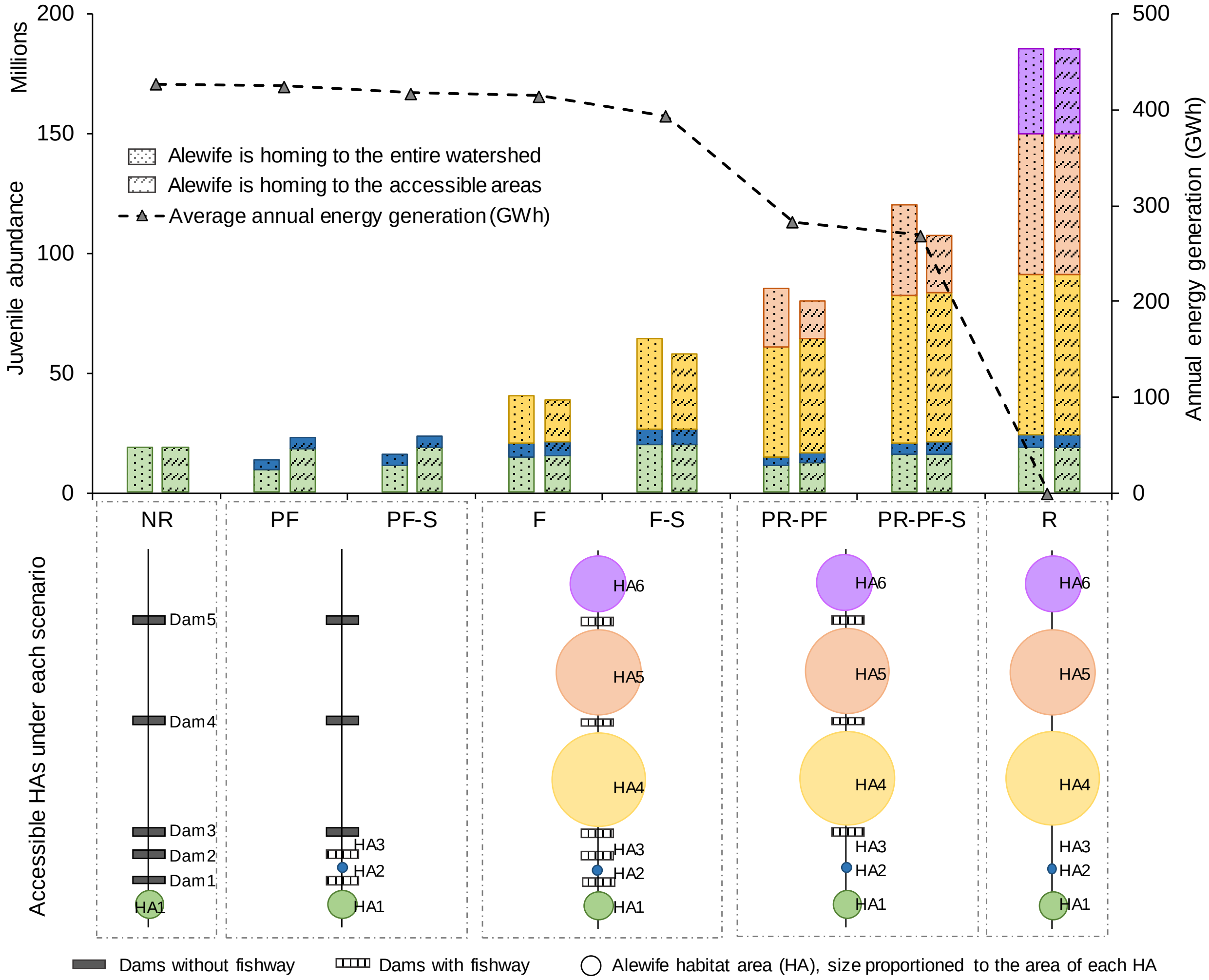


Figure A2. Tradeoffs between energy and Alewife juvenile abundance under different dam management scenarios. Bars filled with different colors are spawner abundance in different HAs. Stabilized spawner abundance of the two dispersal rules are shown as bars filled with dots (homing to the entire basin) and slashes (homing to the accessible areas).

The influence of dams' upstream and downstream passage rates on juvenile abundance was shown in Figure A3. When upstream passage rate is lower than 60%, improvement of both upstream and downstream passage efficiency has a marginal effect on juvenile restoration. The maximum juvenile abundace under these conditions is one-time larger than the impassable condition (0% upstream pass rate for all five dams). Further fish (juvenile) restoration is not possible unless enhance the performance of fishway upstream passage efficiency. When upstream passage rate is larger than 60%, juvenile population is highly influenced by the downstream survival rate. Under a relatively low downstream survival rate of less than 60%, juvenile population is easily to crash due to the extremely low population of spawners. However, if downstream survival rate is higher than 70%, increase upstream passage rate could widely increase juvenile population.

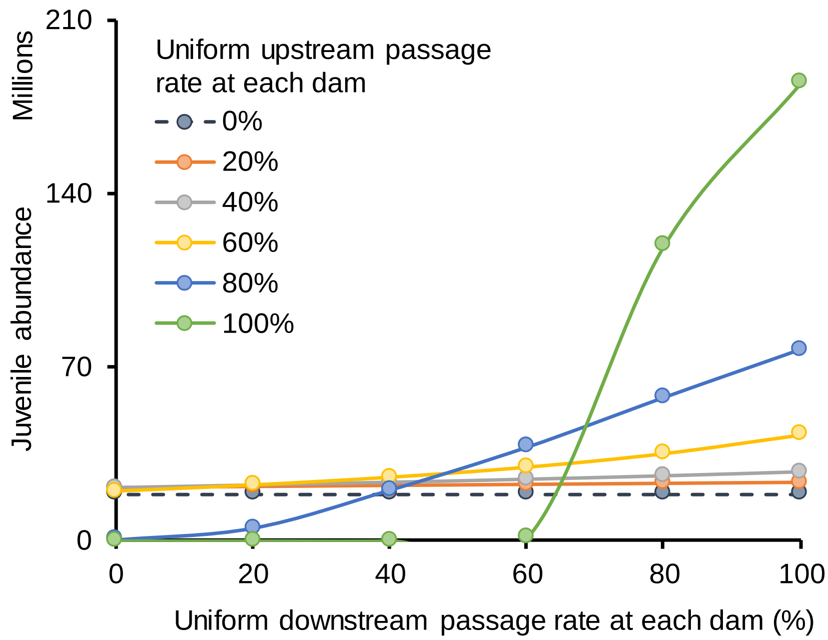


Figure A3. Alewife juvenile abundance in the Penobscot River under various scenarios of upstream fishway passage rate and downstream survival rate. The colored lines correspond to various levels of upstream passage rate at all five dams.

The absolute sensitivity index of juvenile abundance in response to changes of model input parameters in the interval [−0.9, −0.1] and [0.1, 0.9] is shown in Figure A4. Juveniles were sensitive to five input parameters, including ocean mortality, spawning mortality, fecundity slope, the size of habitat area, and asymptotic recruitment level, regardless of percentage of changes. In addition, juveniles were sensitive to −90% to −10% decrease and 10% to 50% increase of alpha and sex ratio. Other two parameters, including fishing mortality and fishway passage rate, in a shift of −10% decrease or any percentage of increase could also cause big influence on juvenile abundance. A comparison between spawners' and juveniles' sensitivity analysis shown that juveniles were sensitive to fishway passage rate, but not sensitive to turbine mortalities.

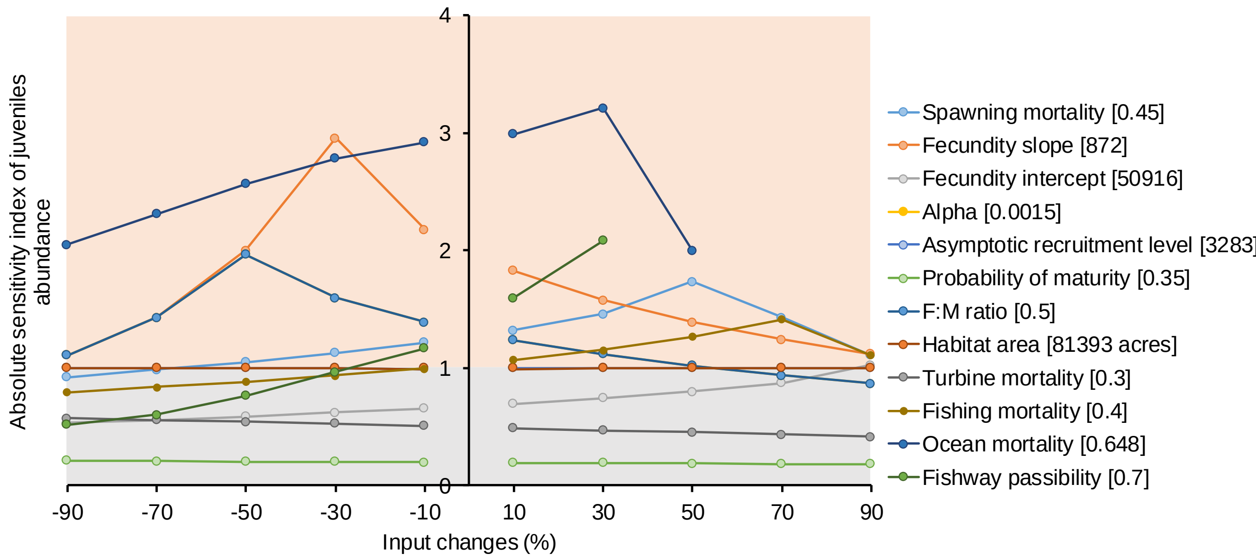


Figure A4. Sensitivity analysis index of alewife juvenile abundance. Outputs of parameters distributed in the light orange shadow are considered highly sensitive, while those distributed in the light grey shadow are not. Numbers in the bracket represent the default value of each input parameter.

APPENDIX B: SUPPORTING INFORMATION FOR CHAPTER 3

Section B1. Modelled life stages, description, governing equation, and parameter values of four fish species

Table B1. Life cycle stages, governing equation, and associated parameters' values of four studied fish species

Life stages	Governing equation					
Egg deposition ($E_{HAj,t,a}$) is the number of eggs produced in each habitat area (*HA*) for a given year *t* on the *a*th day. It was a function of females that survived to spawn in that area and their fecundity. This stage is only suit for alewife, American shad, and Atlantic salmon.	$E_{HA_j,t,a} = \sum_{i=i_{min}}^{i_{max}} (S_{HA_j,i,t,a} \times r_{F:M} \times \varphi \times F_i)$					
	Parameters		**Value or relationship**			
			Alewife	**American shad**	**Atlantic salmon**	**Sea lamprey**
	a	The day of spawners deposit eggs and migrate downstream each year	140*	180 (Castro-Santos and Letcher, 2010)	270 (Legault, 2005)	190 (Beamish and Potter, 1975)
	i_{min}	Minimum age reaching sexual maturity	3	4 (Bailey and Zydlewski, 2013)	4	5
	i_{max}	Maxmum age reaching sexual maturity	6 (Messieh, 1977)	8 (Bailey and Zydlewski, 2013)	9	8
	$r_{F:M}$	Female to male ratio	0.5 (Barber et al., 2018)	0.5 (Bailey and Zydlewski, 2013)	0.5 (Legault, 2004; Legault, 2005)	0.47 (Beamish and Potter, 1975; Howe et al., 2012)
	φ	Probability of spawning	0.95 (Barber et al., 2018)	0.9	1	1
	F_i	Fecundity of age-i spawners	F_i is linearly related to its mass, W_i **$F_i = \mu W_i - v$** Where, μ is fecundity slope, 872; v is fecundity intercept, 50916; the value of W_3 to W_6 is 144, 186, 209, and 244 g (Barber et al., 2018; Fisheries and Oceans Canada et al., 1981-2016).	F_i is exponentially related to its length, L_i **$F_i = 10^{\mu \times L_i + v}$** Where, μ is fecundity slope, 0.045; v is fecundity intercept, 2.2; the value of L_4 to L_8 is 47, 52, 57, 60, and 61 cm (Bailey and Zydlewski, 2013).	The value of F_4 and F_5 are 3040 and 7560, respectively. The value of F_6 to F_9 are 20000	NA
Recruit production ($R_{HAj,t,b}$) is the number of recruits at *HAj* for a given	Alewife and Atlantic salmon are adopted the Berverton-Holt spawner-recruit curve: $R_{HA_j,t,b} = \dfrac{\alpha \times E_{HA_j,t,a}}{1 + \dfrac{\alpha \times E_{HA_j,t,a}}{A_j \times R_{asy}}}$					

<table>
<tr>
<td rowspan="7">year t on the day of downstream migration.
It was modeled as a density-dependent process which was mainly determined by the carrying capacity of spawning and rearing grounds.</td>
<td colspan="6">American shad is adopted the Ricker spawner-recruit curve:
$$R_{HA_j,t,b} = \varepsilon \times E_{HA_j,t,a} \times e^{\alpha \times (1-\frac{E_{HA_j,t,a}}{A_j \times R_{asy}})}$$
Sea lamprey is adopted the Ricker spawner-recruit curve:
$$R_{HA_j,t,b} = \varepsilon \times S_{HA_j,t,a} \times e^{\alpha \times (1-\frac{S_{HA_j,t,a}}{A_j \times R_{asy}})}$$</td>
</tr>
<tr><td colspan="2" rowspan="2">Parameters</td><td colspan="4">Value</td></tr>
<tr><td>Alewife</td><td>American shad</td><td>Atlantic salmon</td><td>Sea lamprey</td></tr>
<tr><td>b</td><td>The day of recruit downstream migration each year</td><td>230</td><td>270 (Greene et al., 2009)</td><td>150 (Legault, 2005)</td><td>330 (Beamish and Potter, 1975)</td></tr>
<tr><td>α</td><td>Maximum reproductive rate</td><td>0.0015 (Barber et al., 2018)</td><td>0.003 (Bailey and Zydlewski, 2013)</td><td>0.1</td><td>12.8 (Dawson and Jones, 2009)</td></tr>
<tr><td>R_{asy}</td><td>Asymptotic recruitment level</td><td>3283 (age-0/acre) (Barber et al., 2018)</td><td>213.7 (age-0/100 m^2) (Bailey and Zydlewski, 2013)</td><td>10 (age-3/100 m^2) (Nieland et al., 2013)</td><td>80.8 (age-0/100 m^2) (Dawson and Jones, 2009)</td></tr>
<tr><td>ε</td><td>Constant variable in recruitment curve</td><td>NA</td><td>0.0007 (Bailey and Zydlewski, 2013)</td><td>NA</td><td>0.0003 (Howe et al., 2012)</td></tr>
<tr><td></td><td>A_j</td><td>Habitat area size in HA_i</td><td>The size of A_1 to A_6 are 7963, 2379, 0, 29506, 25865, and 15680 acres, respectively (TNC, 2016).</td><td>The size of A_1 to A_6 are 32250, 15050, 2150, 88150, 58050, and 19350 (100 m^2), respectively.</td><td>The size of A_1 to A_6 are 22528, 6865, 4, 102031, 56450, and 128537 (100 m^2), respectively (Nieland et al., 2015).</td><td>The size of A_1 to A_6 are the same as Salmon's (Nieland et al., 2015).</td></tr>
<tr>
<td rowspan="2">Juveniles entering ocean ($R_{ocean,t,c}$) is the number of recruits entering ocean after living a certain period in the freshwater area and successfully pass a series of dams along their migration corridor.</td>
<td colspan="6">Recruits of alewife, American shad, and Atlantic salmon were assumed to migrate seaward immediately after they were produced. The number of recruits successfully entering ocean is determined by the cumulative turbine mortality.
$$R_{ocean,t,c} = \sum_{j=1}^{6} R_{HA_j,t,b} \times \prod_{k=1}^{j-1}(1-\frac{Q_{tb_k,t,c}}{Q_{dam_k,t,c}} \times M_{tb})$$
Recruits of sea lamprey will further live in the freshwater for around 3 to 6 years to grow to transformers, when they start to migrate downstream. The number of sea lamprey recruits that successfully entering ocean is mainly determined by survivals during a certain period in the freshwater as well as the cumulative turbine mortality.
$$R_{ocean,t,c} = \sum_{j=1}^{6}\sum_{n=3}^{6} R_{HA_j,(t-n),b} \times e^{-\theta_n M_{fw}} \times T_n \times \prod_{k=1}^{j-1}(1-\frac{Q_{tb_k,t,c}}{Q_{dam_k,t,c}} \times M_{tb})$$</td>
</tr>
<tr><td colspan="2">Parameters</td><td colspan="4">Value</td></tr>
</table>

			Alewife	American shad	Atlantic salmon	Sea lamprey
	c	The day of recruits enter ocean each year	240	280	160	340
	$Q_{tb_k,t,c}$ and $Q_{dam_k,t,c}$ are the turbine and the total water flow rate of Dam *k* (*k* =1-5) in year *t* on the c^{th} day, respectively, m^3/d.					
	M_{tb}	Turbine mortality for juveniles and adults	0.1	0.1	0.1	0.1
	n	The period of lamprey living in the freshwater	NA			3 to 6 years
	Θ_n	The period between recruit production and transformer downstream migration	NA			$\theta_n = \frac{n \times 365 + 140}{365}$
	M_{fw}	Instantaneous annual freshwater mortality rate	NA			0.1 (Howe et al., 2012; Zerrenner, 2001)
	T_n	Transform probability between age-(*n-1)* and age-*n* lamprey	NA			The value of T_3 to T_6 are 0.07, 0.86, 0.9 and 1, respectively (Howe et al., 2012).
Growth and maturity of fish in the ocean	For alewife, American shad, and Atlantic salmon, adults in the ocean in year *t* on the d^{th} day include immature fish and mature fish. **Immature fish (*$NS_{i,t,d}$*)** remains in the ocean, and their abundance was calculated by projecting forward by applying an annual ocean mortality rate on the d^{th} day every year, and the probability of maturation at each age. $NS_{0,t,d}$ is assumed to be equal to recruits entering ocean, $R_{ocean,t,c}$. $$NS_{i,t,d} = NS_{i-1,t-1,d} \times e^{-M_{ocean}} \times (1 - m_i)$$ **Mature fish (*$S_{i,t,d}$*)** includes the first-time spawners, $S_{i,t,0,d}$, and the repeat spawners (exclude sea lamprey), $S_{i,t,p,d}$. The repeat spawners have spawned at least one time and are subject to subject to natural (i.e., predation, delayed migration, or senescence), fishing (both commercial and recreational), and other anthropogenic (i.e., turbine) mortalities prior to their next spawning run. $$S_{i,t,d} = S_{i,t,0,d} + \sum_p S_{i,t,p,d}$$ $$S_{i,t,0,d} = NS_{i-1,t-1,d} \times e^{-M_{ocean}} \times m_i$$					

$$S_{i+1,t+1,p+1,d} = \left(\sum_{j=1}^{6} S_{HA_j,i,t,p,a} \times \prod_{k=1}^{j-1}\left(1 - \frac{Q_{tb_k,t,c}}{Q_{dam_k,t,c}} \times M_{tb}\right)\right) \times e^{-\partial M_{ocean}}$$

For sea lamprey, transformers entering ocean become mature after 2 years. Since lamprey die after spawning, thus all mature lamprey in the ocean are the first-time spawners which is calculated as follows.

$$S_{t,d} = R_{ocean,(t-2),c} \times e^{-\partial M_{ocean}}$$

Parameters		**Value**			
		Alewife	**American shad**	**Atlantic salmon**	**Sea lamprey**
d	The day of mature fish upstream migration each year	120	160 (Greene et al., 2009; Grote et al., 2014)	160 (Legault, 2005)	160 (Beamish and Potter, 1975)
M_{ocean}	Instantaneous annual ocean mortality rate	0.648 (Barber et al., 2018)	0.38 (ASMFC, 2007; Bailey and Zydlewski, 2013)	0.9 (Legault, 2004; Legault, 2005)	1.39 (Howe et al., 2012; Zerrenner, 2001)
m_i	Mature probability between age-(*i*-*1*) and age-*i*	The value of m_3 to m_6 are 0.35, 0.51, 0.96, and 0.96, respectively (Gibson and Myers, 2003).	The value of m_4 to m_8 are 0.2, 0.25, 0.61, 0.86, and 0.96 respectively (ASMFC, 2007; Bailey and Zydlewski, 2013).	For first-time spawners, the value of one-sea-winter (SW1) salmon is 0.02, two-sea-winter salmon is 0.94. For repeat spawners (known as kelt), the value is 0.5. (Legault, 2004; Legault, 2005)	NA
∂	The period of fish living in the ocean out of 365 days	335/365	335/365	610/365	550/365

Spawning runs ($\sum_{j=1}^{6} S_{HA_j,t,a}$) is the total number of spawners reaching the suitable habitat areas in the basin for a given year *t* on the *a*^th^ day. It was determined by fishing and spawning mortalities.

For alewife, American shad, and Atlantic salmon:

$$\sum_{j=1}^{6} S_{HA_j,t,a} = \sum_{i=i_{min}}^{i_{max}} S_{i,t,d} \times (1 - M_{fishing}) \times (1 - M_{spawn})$$

For sea lamprey:

$$\sum_{j=1}^{6} S_{HA_j,t,a} = S_{t,d} \times (1 - M_{fishing}) \times (1 - M_{spawn})$$

The value of $S_{HAj,t,a}$ was determined by the cumulative upstream passage rate of dams downstream of HA_j as well as a dispersal rule. Here, we included the long-term blockage effect of dams that restricts alewives' motivation to seek habitats that were suitable for spawning but no longer accessible. The following equations calculate this dispersal rule.

If $\frac{A_j}{A} < D_{HA_j}$,

$$S_{HA_j,t,a} = \left(\frac{A_j}{A} + \left(D_{HA_j} - \frac{A_j}{A}\right) \times (1 - P_j)\right) \times \sum_{j=1}^{j=6} S_{HA_j,t,a}$$

If $\frac{A_j}{A} \geq D_{HA_j}$,

$$S_{HA_j,t,a} = D_{HA_j} \times \sum_{j=1}^{j=6} S_{HA_j,t,a}$$

D_{HA_j} is a dispersal factor (D_{HA_1} = 1), which is calculated as follows,

$$D_{HA_j} = (D_{HA_{j-1}} - \frac{A_{j-1}}{A}) \times P_{j-1}$$

Parameters		**Value**			
		Alewife	**American shad**	**Atlantic salmon**	**Sea lamprey**
$M_{fishing}$	Medium interval fishing mortality	0.4 (Barber et al., 2018)	0	0	0
M_{spawn}	Average interval spawning mortality rate	0.45 (Barber et al., 2018; Durbin et al., 1979; Kissil, 1974)	0.3 (Bailey and Zydlewski, 2013)	0.473 Maynard, Izzo, and Zydlewski, kelt telemetry	1
A	The sum of *HAs* that are accessible by fish species	Varies depending on the passage rate at each dam			
P_j	The passage rate of the j^{th} dam	Its value is provided in Table 1 of the main text			

Section B2. Sensitivity analysis

Table B2. The parameters, values, and ranges of sensitivity analysis in the age-structured fish population model

Alewife				American shad				Atlantic salmon				Sea lamprey			
Parameters	Values	Ranges		Parameters	Values	Ranges		Parameters	Values	Ranges		Parameters	Values	Ranges	
Female to male ratio	0.5	0.4	0.6	Female to male ratio	0.5	0.4	0.6	Female to male ratio	0.5	0.4	0.6	Female to male ratio	0.47	0.376	0.564
Probability of spawning	0.95	0.76	1	Probability of spawning	0.9	0.72	1	SW1 fecundity	3040	2432	3648	Max reproductive rate	12.8	10.24	15.36
Fecundity slope	872	697.6	1046.4	Fecundity slope	0.045	0.036	0.054	SW2 fecundity	7560	6048	9072	Asymptotic recruitment level	80.8	64.64	96.96
Fecundity intercept	50916	40732.8	61099.2	Fecundity intercept	2.2	1.76	2.64	Kelt fecundity	20000	16000	24000	Constant variable in	0.0003	0.00024	0.00036

												recruitment curve			
Max reproductive rate	0.0015	0.0012	0.0018	Max reproductive rate	0.003	0.0024	0.0036	Max reproductive rate	0.1	0.08	0.12	Annual freshwater mortality	0.1	0.08	0.12
Asymptotic recruitment level	3283	2626.4	3939.6	Asymptotic recruitment level	213.7	170.96	256.44	Asymptotic recruitment level	10	8	12	Age-3 transform probability	0.07	0.056	0.084
Annual ocean mortality	0.648	0.5184	0.7776	Constant variable in recruitment curve alpha	0.0007	0.00056	0.00084	Annual ocean mortality	0.9	0.72	1.08	Age-4 transform probability	0.86	0.688	1
Age-3 mature probability	0.35	0.28	0.42	Annual ocean mortality	0.38	0.304	0.456	SW1 mature probability	0.02	0.016	0.024	Age-5 transform probability	0.9	0.72	1
Age-4 mature probability	0.51	0.408	0.612	Age-4 mature probability	0.2	0.16	0.24	SW2 mature probability	0.94	0.752	1	Annual ocean mortality	1.39	1.112	1.668
Age-5 mature probability	0.96	0.768	1	Age-5 mature probability	0.25	0.2	0.3	Kelt mature probability	0.5	0.4	0.6	Pool-and-weir passage rate	0.35	0.1	0.9
Age-6 mature probability	0.96	0.768	1	Age-6 mature probability	0.61	0.488	0.732	Spawning mortality	0.473	0.3784	0.5676	Denil passage rate	0.2	0.1	0.9
Fishing mortality	0.4	0.32	0.48	Age-7 mature probability	0.86	0.688	1	Pool-and-weir passage rate	0.72	0.1	0.9	Fish lift passage rate	0.6	0.1	0.9
Spawning mortality	0.45	0.36	0.54	Age-8 mature probability	0.96	0.768	1	Denil passage rate	0.22	0.1	0.9	Total habitat areas	316415	253132	379698
Pool-and-weir passage rate	0.46	0.1	0.9	Spawning mortality	0.3	0.24	0.36	Fish lift passage rate	0.36	0.1	0.9				
Denil passage rate	0.82	0.1	0.9	Pool-and-weir passage rate	0.15	0.1	0.9	Total habitat areas	316415	253132	379698				
Fish lift passage rate	0.7	0.1	0.9	Denil passage rate	0.61	0.1	0.9								
Total habitat areas	81393	65114.4	97671.6	Fish lift passage rate	0.29	0.1	0.9								
				Total habitat areas	215000	172000	258000								

Table B3. The parameters, values, and ranges of sensitivity analysis in the cost model

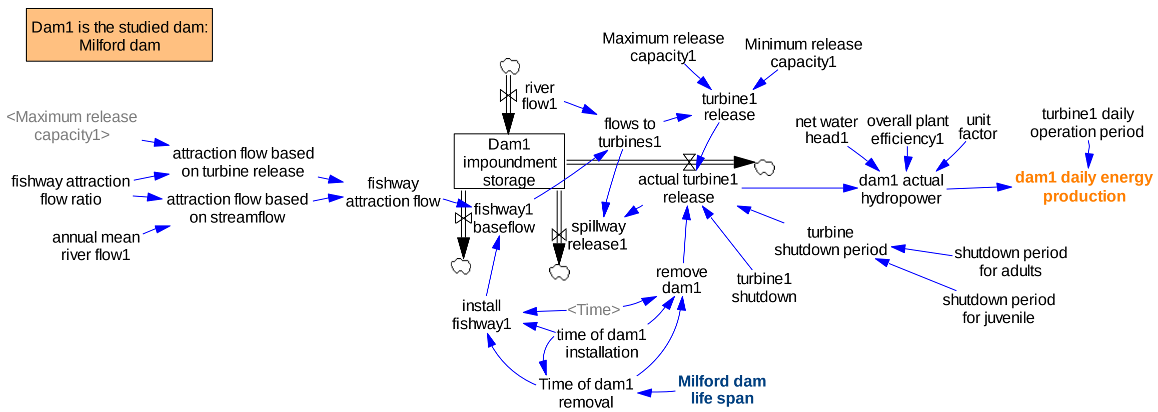
Parameters	Values	Ranges	
Dam removal cost per vertical height	0.173	0.002	0.7
Pool-and-weir capital cost per vertical meter	0.178	0.007	0.178
Denil capital cost per vertical meter	0.190	0.019	0.361
Fish lift capital cost per vertical meter	0.237	0.05	0.44
Fishway annual O&M cost	2% of capital cost	1% of capital cost	5% of capital cost

APPENDIX C: SUPPORTING INFORMATION FOR CHAPTER 4

The energy-fish model was built in the Vensim® DSS. It is composed by four sub-models, including energy generation model (Section C1), age-structured fish population model (Section C2), upstream migration model (Section C3), and downstream migration model (Section C4). The complete version of the model structure and its equations of each sub-model are provided below.

Section C1. Energy generation model

Model structure



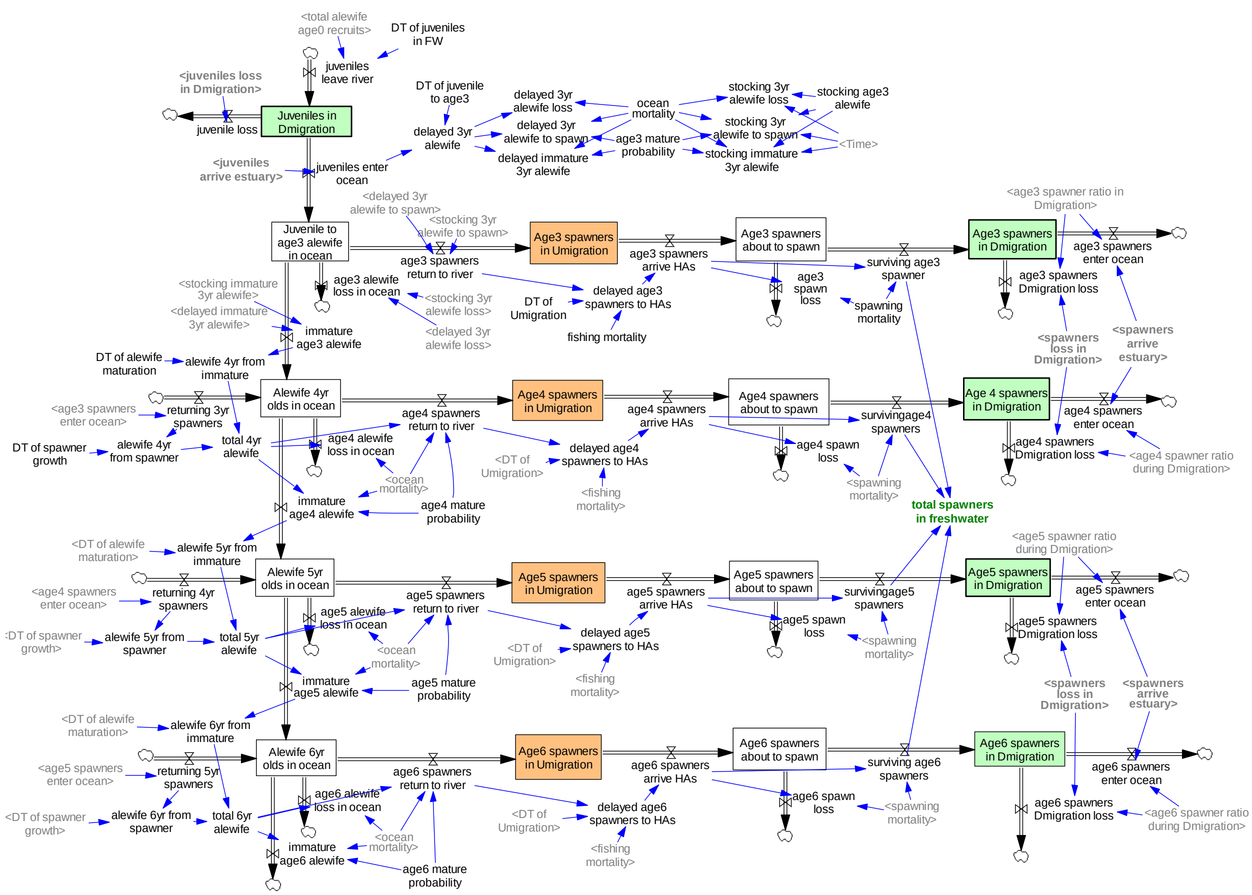
Equations embedded in the model

Actual turbine1 release = IF THEN ELSE (remove dam1 = 1, 0, IF THEN ELSE (turbine1 shutdown = 1 :AND: turbine shutdown period = 1, 0, turbine1 release))
Annual mean river flow1 = 2.50891e+007
Attraction flow based on streamflow = fishway attraction flow ratio * annual mean river flow1
Attraction flow based on turbine release = fishway attraction flow ratio * maximum release capacity1
Dam1 actual hydropower = overall plant efficiency1 * net water head1 * actual turbine1 release * 1000 * 9.81/1e+006 * unit factor
Dam1 daily energy production = dam1 actual hydropower * turbine1 daily operation period
Dam1 impoundment storage = INTEG (river flow1 - fishway1 baseflow - spillway release1 - actual turbine1 release, 1.76696e+007)
Fishway attraction flow = IF THEN ELSE (attraction flow based on streamflow >= attraction flow based on turbine release, attraction flow based on streamflow, attraction flow based on turbine release)
Fishway attraction flow ratio = 0.05
Fishway1 baseflow = IF THEN ELSE (install fishway1 = 1, fishway attraction flow, 0)
Flows to turbines1 = river flow1 - fishway1 baseflow
Install fishway1 = IF THEN ELSE (Time >= time of dam1 installation :AND: Time <= Time of dam1 removal, 1, 0)
Maximum release capacity1 = 1.72e+007
Milford dam life span = 30
Minimum release capacity1 = 6.9e+006
Net water head1 = 5.8

Overall plant efficiency1 = 0.85
Remove dam1 = IF THEN ELSE (Time < time of dam1 installation :OR: Time >= Time of dam1 removal, 1, 0)
River flow1 = GET XLS DATA ('River discharge from USGS.xlsx', 'Daily discharge', 'A', 'G3')
Shutdown period for adults = PULSE TRAIN (141, 10, 365, 73000)
Shutdown period for juvenile = PULSE TRAIN (231, 10, 365, 73000)
Spillway release1 = flows to turbines1 - actual turbine1 release
Time of dam1 installation = 25550
Time of dam1 removal = time of dam1 installation + Milford dam life span * 365
Turbine shutdown period = shutdown period for adults + shutdown period for juvenile
Turbine1 daily operation period = 24
Turbine1 release = IF THEN ELSE (flows to turbines1 >= maximum release capacity1, maximum release capacity1, IF THEN ELSE (flows to turbines1 < minimum release capacity1, 0, flows to turbines1))
Turbine1 shutdown = 1
Unit factor = 1/24/3600

Section C2. Age-structured fish population model

Model structure



Equations embedded in the model

Age3 alewife loss in ocean = delayed 3yr alewife loss + stocking 3yr alewife loss

Age3 mature probability = 0.35
Age3 spawn loss = age3 spawners arrive HAs * spawning mortality
Age3 spawner ratio in Dmigration = DELAY FIXED (age3 spawner ratio, DT of Dmigration, 0)
Age3 spawners about to spawn = INTEG (age3 spawners arrive HAs - age3 spawn loss - surviving age3 spawner, 0)
Age3 spawners arrive HAs = delayed age3 spawners to HAs
Age3 spawners Dmigration loss = spawners loss in Dmigration * age3 spawner ratio in Dmigration

Age3 spawners enter ocean = spawners arrive estuary * age3 spawner ratio in Dmigration
Age3 spawners in Dmigration = INTEG (surviving age3 spawner - age3 spawners enter ocean - age3 spawners Dmigration loss, 0)
Age3 spawners in Umigration = INTEG (age3 spawners return to river - age3 spawners arrive HAs, 0)
Age3 spawners return to river = delayed 3yr alewife to spawn + stocking 3yr alewife to spawn
Age4 alewife loss in ocean = total 4yr alewife * (1 - EXP (-0.92 * ocean mortality))
Age4 mature probability = 0.51
Age4 spawn loss = age4 spawners arrive HAs * spawning mortality
Age4 spawner ratio during Dmigration = DELAY FIXED (age4 spawner ratio, DT of Dmigration, 0)
Age4 spawners about to spawn = INTEG (age4 spawners arrive HAs - age4 spawn loss - survivingage4 spawners, 0)
Age4 spawners arrive HAs = delayed age4 spawners to HAs
Age4 spawners Dmigration loss = spawners loss in Dmigration * age4 spawner ratio during Dmigration
Age4 spawners enter ocean = spawners arrive estuary * age4 spawner ratio during Dmigration
Age 4 spawners in Dmigration = INTEG (survivingage4 spawners - age4 spawners enter ocean - age4 spawners Dmigration loss, 0)
Age4 spawners in Umigration = INTEG (age4 spawners return to river - age4 spawners arrive HAs, 0)
Age4 spawners return to river = total 4yr alewife * EXP (-0.92 * ocean mortality) * age4 mature probability
Age5 alewife loss in ocean = total 5yr alewife * (1 - EXP (-0.92 * ocean mortality))
Age5 mature probability = 0.96
Age5 spawn loss = age5 spawners arrive HAs * spawning mortality
Age5 spawner ratio during Dmigration = DELAY FIXED (age5 spawner ratio, DT of Dmigration, 0)
Age5 spawners about to spawn = INTEG (age5 spawners arrive HAs - age5 spawn loss - survivingage5 spawners, 0)
Age5 spawners arrive HAs = delayed age5 spawners to HAs
Age5 spawners Dmigration loss = spawners loss in Dmigration * age5 spawner ratio during Dmigration
Age5 spawners enter ocean = spawners arrive estuary * age5 spawner ratio during Dmigration
Age5 spawners in Dmigration = INTEG (survivingage5 spawners - age5 spawners enter ocean - age5 spawners Dmigration loss, 0)
Age5 spawners in Umigration = INTEG (age5 spawners return to river - age5 spawners arrive HAs, 0)

Age5 spawners return to river = total 5yr alewife * EXP (-0.92 * ocean mortality) * age5 mature probability
Age6 alewife loss in ocean = total 6yr alewife * (1 - EXP (-0.92 * ocean mortality))
Age6 mature probability = 0.96
Age6 spawn loss = age6 spawners arrive HAs * spawning mortality
Age6 spawner ratio during Dmigration = DELAY FIXED (age6 spawner ratio, DT of Dmigration, 0)
Age6 spawners about to spawn = INTEG (age6 spawners arrive HAs - age6 spawn loss - surviving age6 spawners, 0)
Age6 spawners arrive HAs = delayed age6 spawners to HAs
Age6 spawners Dmigration loss = age6 spawner ratio during Dmigration * spawners loss in Dmigration
Age6 spawners enter ocean = spawners arrive estuary * age6 spawner ratio during Dmigration
Age6 spawners in Dmigration = INTEG (surviving age6 spawners - age6 spawners enter ocean - age6 spawners Dmigration loss, 0)
Age6 spawners in Umigration = INTEG (age6 spawners return to river - age6 spawners arrive HAs, 0)
Age6 spawners return to river = total 6yr alewife * EXP (-0.92 * ocean mortality) * age6 mature probability
Alewife 4yr from immature = DELAY FIXED (immature age3 alewife, DT of alewife maturation, 0)
Alewife 4yr from spawner = DELAY FIXED (returning 3yr spawners, DT of spawner growth, 0)

Alewife 4yr olds in ocean = INTEG (immature age3 alewife + returning 3yr spawners - age4 spawners return to river - age4 alewife loss in ocean - immature age4 alewife, 0)
Alewife 5yr from immature = DELAY FIXED (immature age4 alewife, DT of alewife maturation, 0)
Alewife 5yr from spawner = DELAY FIXED (returning 4yr spawners, DT of spawner growth, 0)

Alewife 5yr olds in ocean = INTEG (immature age4 alewife + returning 4yr spawners - age5 alewife loss in ocean - age5 spawners return to river - immature age5 alewife, 0)
Alewife 6yr from immature = DELAY FIXED (immature age5 alewife, DT of alewife maturation, 0)
Alewife 6yr from spawner = DELAY FIXED (returning 5yr spawner, DT of spawner growth, 0)
Alewife 6yr olds in ocean = INTEG (returning 5yr spawners + immature age5 alewife - age6 alewife loss in ocean - age6 spawners return to river - immature age6 alewife, 0)
Delayed 3yr alewife = DELAY FIXED (juveniles enter ocean, DT of juvenile to age3, 0)
Delayed 3yr alewife loss = delayed 3yr alewife * (1 - EXP (- ocean mortality * 945/365))
Delayed 3yr alewife to spawn = delayed 3yr alewife * age3 mature probability * EXP (- ocean mortality * 945/365)
Delayed age3 spawners to HAs = DELAY FIXED (age3 spawners return to river * (1 - fishing mortality), DT of Umigration, 0)
Delayed age4 spawners to HAs = DELAY FIXED (age4 spawners return to river * (1 - fishing mortality), DT of Umigration, 0)

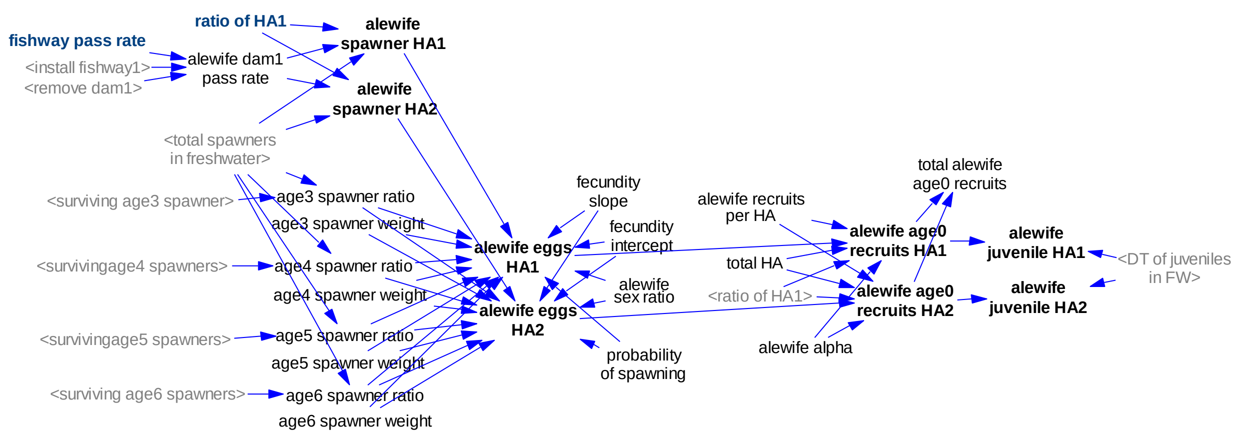
Delayed age5 spawners to HAs = DELAY FIXED (age5 spawners return to river * (1 - fishing mortality), DT of Umigration, 0)
Delayed age6 spawners to HAs = DELAY FIXED (age6 spawners return to river * (1 - fishing mortality), DT of Umigration, 0)
Delayed immature 3yr alewife = delayed 3yr alewife * (1 - age3 mature probability) * EXP (-ocean mortality * 945/365)
DT of alewife maturation = 365
DT of juvenile to age3 = 975
DT of juveniles in FW = 90
DT of spawner growth = 335
DT of Umigration = 20
Fishing mortality = 0.4
Immature age3 alewife = delayed immature 3yr alewife + stocking immature 3yr alewife
Immature age4 alewife = total 4yr alewife * EXP (-0.92 * ocean mortality) * (1 - age4 mature probability)
Immature age5 alewife = total 5yr alewife * EXP (-0.92 * ocean mortality) * (1 - age5 mature probability)
Immature age6 alewife = total 6yr alewife * EXP (-0.92 * ocean mortality) * (1 - age6 mature probability)
Juvenile loss = juveniles loss in Dmigration
Juvenile to age3 alewife in ocean = INTEG (juveniles enter ocean - age3 alewife loss in ocean - immature age3 alewife - age3 spawners return to river, 1e+008)
Juveniles arrive estuary = juveniles at HA1 + juveniles leave dam1
Juveniles enter ocean = juveniles arrive estuary
Juveniles in Dmigration = INTEG (juveniles leave river - juvenile loss - juveniles enter ocean, 0)

Juveniles leave river = DELAY FIXED (total alewife age0 recruits, DT of juveniles in FW, 0)
Juveniles loss in Dmigration = juveniles loss at dam1
Ocean mortality = 0.648
Returning 3yr spawners = age3 spawners enter ocean
Returning 4yr spawners = age4 spawners enter ocean
Returning 5yr spawners = age5 spawners enter ocean
Spawners arrive estuary = survived spawners at HA1 + spawners leave dam1
Spawners loss in Dmigration = spawners loss at dam1
Spawning mortality = 0.45
Stocking 3yr alewife loss = IF THEN ELSE (Time = 120, stocking age3 alewife * (1 - EXP (-ocean mortality * 975/365)), 0)
Stocking 3yr alewife to spawn = IF THEN ELSE (Time = 120, stocking age3 alewife * EXP (-ocean mortality * 975/365) * age3 mature probability, 0)
Stocking age3 alewife = 1e+008
Stocking immature 3yr alewife = IF THEN ELSE (Time = 120, stocking age3 alewife * EXP (-ocean mortality * 975/365) * (1 - age3 mature probability), 0)
Surviving age3 spawner = age3 spawners arrive HAs * (1 - spawning mortality)
Surviving age6 spawners = age6 spawners arrive HAs * (1 - spawning mortality)
Survivingage4 spawners = age4 spawners arrive HAs * (1 - spawning mortality)
Survivingage5 spawners = age5 spawners arrive HAs * (1 - spawning mortality)

Total 4yr alewife = alewife 4yr from immature + alewife 4yr from spawner
Total 5yr alewife = alewife 5yr from immature + alewife 5yr from spawner
Total 6yr alewife = alewife 6yr from spawner + alewife 6yr from immature
Total alewife age0 recruits = alewife age0 recruits HA1 + alewife age0 recruits HA2
Total spawners in freshwater = surviving age3 spawner + survivingage4 spawners + survivingage5 spawners + surviving age6 spawners
Age3 spawner ratio in Dmigration = DELAY FIXED (age3 spawner ratio, DT of Dmigration, 0)
Age4 spawner ratio during Dmigration = DELAY FIXED (age4 spawner ratio, DT of Dmigration, 0)
Age5 spawner ratio during Dmigration = DELAY FIXED (age5 spawner ratio, DT of Dmigration, 0)
Age6 spawner ratio during Dmigration = DELAY FIXED (age6 spawner ratio, DT of Dmigration, 0)

Section C3. Fish upstream migration model

Model structure



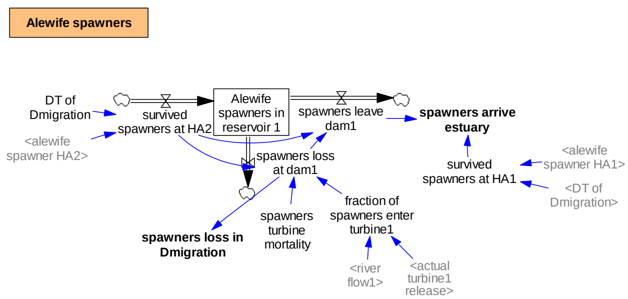
Equations embedded in the model

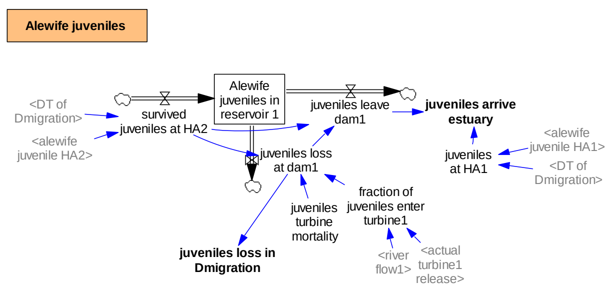
Age3 spawner ratio = IF THEN ELSE (total spawners in freshwater = 0, 0, surviving age3 spawner/total spawners in freshwater)
Age3 spawner weight = 144
Age3 spawners arrive HAs = delayed age3 spawners to HAs
Age4 spawner ratio = IF THEN ELSE (total spawners in freshwater = 0, 0, survivingage4 spawners/total spawners in freshwater)
Age4 spawner weight = 186
Age4 spawners arrive HAs = delayed age4 spawners to HAs
Age5 spawner ratio = IF THEN ELSE (total spawners in freshwater = 0, 0, survivingage5 spawners/total spawners in freshwater)
Age5 spawner weight = 209
Age5 spawners arrive HAs = delayed age5 spawners to HAs
Age6 spawner ratio = IF THEN ELSE (total spawners in freshwater = 0, 0, surviving age6 spawners/total spawners in freshwater)
Age6 spawner weight = 244

Alewife age0 recruits HA1 = IF THEN ELSE (ratio of HA1 = 0, 0, alewife alpha * alewife eggs HA1/(1 + (alewife alpha * alewife eggs HA1/(total HA * ratio of HA1 * alewife recruits per HA))))
Alewife age0 recruits HA2 = IF THEN ELSE ((1 - ratio of HA1) = 0, 0, alewife alpha * alewife eggs HA2/(1 + (alewife alpha * alewife eggs HA2/((1 - ratio of HA1) * total HA * alewife recruits per HA))))
Alewife alpha = 0.0015
Alewife dam1 pass rate = IF THEN ELSE (remove dam1 = 1, 1, IF THEN ELSE (install fishway1 = 1, fishway pass rate 0))
Alewife eggs HA1 = alewife sex ratio * alewife spawner HA1 * probability of spawning * (age3 spawner ratio * (fecundity slope * age3 spawner weight - fecundity intercept) + age4 spawner ratio * (fecundity slope * age4 spawner weight - fecundity intercept) + age5 spawner ratio * (fecundity slope * age5 spawner weight - fecundity intercept) + age6 spawner ratio * (fecundity slope * age6 spawner weight - fecundity intercept))
Alewife eggs HA2 = alewife sex ratio * alewife spawner HA2 * probability of spawning * (age3 spawner ratio * (fecundity slope * age3 spawner weight - fecundity intercept) + age4 spawner ratio * (fecundity slope * age4 spawner weight - fecundity intercept) + age5 spawner ratio * (fecundity slope * age5 spawner weight - fecundity intercept) + age6 spawner ratio * (fecundity slope * age6 spawner weight - fecundity intercept))
Alewife juvenile HA1 = DELAY FIXED (alewife age0 recruits HA1, DT of juveniles in FW, 0)
Alewife juvenile HA2 = DELAY FIXED (alewife age0 recruits HA2, DT of juveniles in FW, 0)
Alewife recruits per HA = 811246
Alewife sex ratio = 0.5
Alewife spawner HA1 = ratio of HA1 * total spawners in freshwater + (1 - ratio of HA1) * total spawners in freshwater * (1 - alewife dam1 pass rate)
Alewife spawner HA2 = IF THEN ELSE ((1 - ratio of HA1) * alewife dam1 pass rate <= 0, 0, (1 - ratio of HA1) * alewife dam1 pass rate * total spawners in freshwater)
DT of juveniles in FW = 90
Fecundity intercept = 50916
Fecundity slope = 872
Fishway pass rate = 0
Install fishway1 = IF THEN ELSE (Time >= time of dam1 installation :AND: Time <= Time of dam1 removal, 1, 0)
Probability of spawning = 0.95
Ratio of HA1 = 0.13
Remove dam1 = IF THEN ELSE (Time < time of dam1 installation :OR: Time >= Time of dam1 removal, 1, 0)
Spawning mortality = 0.45
Surviving age3 spawner = age3 spawners arrive HAs * (1 - spawning mortality)
Surviving age6 spawners = age6 spawners arrive HAs * (1 - spawning mortality)
Survivingage4 spawners = age4 spawners arrive HAs * (1 - spawning mortality)
Survivingage5 spawners = age5 spawners arrive HAs * (1 - spawning mortality)
Total alewife age0 recruits = alewife age0 recruits HA1 + alewife age0 recruits HA2
Total HA = 330
Total spawners in freshwater = surviving age3 spawner + survivingage4 spawners + survivingage5 spawners + surviving age6 spawners

Section C4. Fish downstream migration model

Model structure





Equations embedded in the model

For alewife spawners
Actual turbine1 release = IF THEN ELSE (remove dam1 = 1, 0, IF THEN ELSE (turbine1 shutdown = 1 :AND: turbine shutdown period = 1, 0, turbine1 release))
Alewife spawner HA1 = ratio of HA1 * total spawners in freshwater + (1 - ratio of HA1) * total spawners in freshwater * (1 - alewife dam1 pass rate)
Alewife spawner HA2 = IF THEN ELSE ((1 - ratio of HA1) * alewife dam1 pass rate <= 0, 0, (1 - ratio of HA1) * alewife dam1 pass rate * total spawners in freshwater)
Alewife spawners in reservoir 1 = INTEG (survived spawners at HA2 - spawners leave dam1 - spawners loss at dam1, 0)
DT of Dmigration = 10
Fraction of spawners enter turbine1 = actual turbine1 release / river flow1
River flow1= GET XLS DATA ('River discharge from USGS.xlsx', 'Daily discharge', 'A', 'G3')
Spawners arrive estuary = survived spawners at HA1 + spawners leave dam1
Spawners leave dam1 = survived spawners at HA2 - spawners loss at dam1
Spawners loss at dam1 = survived spawners at HA2 * fraction of spawners enter turbine1 * spawners turbine mortality
Spawners loss in Dmigration = spawners loss at dam1
Spawners turbine mortality = 0.3
Survived spawners at HA1 = DELAY FIXED (alewife spawner HA1, DT of Dmigration, 0)
Survived spawners at HA2 = DELAY FIXED (alewife spawner HA2, DT of Dmigration, 0)

For alewife juveniles
Actual turbine1 release = IF THEN ELSE (remove dam1 = 1, 0, IF THEN ELSE (turbine1 shutdown = 1 :AND: turbine shutdown period = 1, 0, turbine1 release))
Alewife juvenile HA1 = DELAY FIXED (alewife age0 recruits HA1, DT of juveniles in FW, 0)
Alewife juvenile HA2 = DELAY FIXED (alewife age0 recruits HA2, DT of juveniles in FW, 0)
Alewife juveniles in reservoir 1 = INTEG (survived juveniles at HA2 - juveniles leave dam1 - juveniles loss at dam1, 0)
DT of Dmigration = 10
Fraction of juveniles enter turbine1 = actual turbine1 release/river flow1
Juveniles arrive estuary = juveniles at HA1 + juveniles leave dam1
Juveniles at HA1 = DELAY FIXED (alewife juvenile HA1, DT of Dmigration, 0)
Juveniles leave dam1 = survived juveniles at HA2 - juveniles loss at dam1

Juveniles loss at dam1 = survived juveniles at HA2 * fraction of juveniles enter turbine1 * juveniles turbine mortality
Juveniles loss in Dmigration = juveniles loss at dam1
Juveniles turbine mortality = 0.3
River flow1 = GET XLS DATA ('River discharge from USGS.xlsx', 'Daily discharge', 'A', 'G3')
Survived juveniles at HA2 = DELAY FIXED (alewife juvenile HA2, DT of Dmigration, 0)

APPENDIX D: SUPPORTING INFORMATION FOR CHAPTER 5

Table D1. Life cycle greenhouse gas (GHG) emissions of different hydropower projects estimated by previous life cycle assessment studies

NO	Location	Method	Life span (year)	Dam type	Ref.	Installed capacity	GWP at different stages (g CO_2 eq./ kWh)							
							Carbon loss of flooded plant	Construction			O&M	End-of-life		Total
								Manufacturing	Transportation	Construction		Dam removal	Sediment release	
Diversion HPs														
1	Thailand	PCA	20	Weir	(Pascale et al., 2011)	3 kW		√			√	√		52.7
2	UK	PCA	50	Weir	(Gallagher, J. et al., 2015a)	50 kW		8.5	0.3	0.2				8.9
3						100 kW		7.0	0.2	0.3				7.4
4						650 kW		5.4	0.06	0.06				5.5
5	Thailand	PCA	50	Mass concrete weir	(Suwanit and Gheewala, 2011)	1150 kW			5.6	10.6	1.4	0.1		16.5
6						2250 kW								22.7
7						2500 kW								16.3
8						5100 kW								11.0
9						6000 kW								23.0
10	China	PCA	30	Concrete gravity dam	(Pang et al., 2015)	3200 kW			0.2	27.3	0.9	0.0		28.4
11	India	EIO	30	Concrete dam	(Varun et al., 2010)	1000 kW		6.1[a] + 8.5[b]			16.5			31.2
12						2000 kW		5.0[a] + 4.6[b]			9.8			19.4
13	India	EIO	30	Canal-based	(Varun et al., 2008; Varun et al., 2012)	50 kW		23.1[a] + 14.8[b]			37.0			74.9
14						100 kW		11.8[a] + 12.0[b]			31.6			55.4
15						3000 kW		8.1[a] + 8.4[b]			18.7			35.3
16	India 23	EIO	30	canal-based	(Varun et al., 2010)	250 kW		9.4[a] + 8.5[b]			17.5			35.4
17						400 kW		9.7[a] + 7.7[b]			16.5			33.9
18						1000 kW		6.8[a] + 9.9[b]			15.5			32.2
19	Japan	Hybrid		Concrete dam	(Hondo, 2005)	10 MW		9.4			1.9			11.3
Reservoir-based HPs														

20	India	EIO	30	Concrete dam	(Varun et al., 2010)	30 MW		1.4[a] + 4.7[b]			5.8			11.9
21	China	EIO	50	Rock-fill concrete	(Zhang et al., 2007)	44 MW		12		11	21 (R)			44
22			100	Double arch concrete		3600 MW		1.6		0.8	3.7 (R)			6.1
25	US	EIO	100	Concrete arch dam	(Pacca, 2007)	1296 MW		2.3			34-77 (R)		40-148	76.3-227.3
26	US	EIO	20	Concrete arch dam	(Pacca and Horvath, 2002)	1296 MW	3.6	4.5			27.3-54.5 (R)			35.4-62.6
23	China	Hybrid	50	earth-core rock dam	(Zhang, S. et al., 2015)	5850 MW		2.6	0.2	0.4	5.22 (R)			8.4
24			50	concrete gravity dam		5850 MW		4.8	0.2	0.3	5.7 (R)			11.1
27	Brazil	PCA	100		(Ribeiro and da Silva, 2010)	12600MW		√			√			4.3
28	China	Hybrid	50	Rock-filled concrete dam	(Liu et al., 2013)			√	√	√	√			0.4 $kgCO_2$ eq./kg concrete cast
29				Concrete dam				√	√	√	√			0.2 $kgCO_2$ eq./kg concrete cast
Pumped storage HPs														
30	Belgium	PCA	150	(Pumped-storage HP)	(Oliveira et al., 2015)	22 MW		5.0			20-640			25-645
31	US	EIO	60	Earth/rock-fill dam;	(Denholm and Kulcinski, 2004)	840 MW		4.0			1.8 (R)	1.0		5.6
River in-stream HPs														
32	UK	PCA	30	No dam; (Hydrokinetic)	(Gallagher, John et al., 2015)	15 kW		√			√			2.1
33						90 kW		√			√			4.4
34						140 kW		√			√			2.8
35	US	PCA	100	No dam; (Hydrokinetic)	(Miller et al., 2011)	6 W		√			√	√		4.8

Note: R: Include GHG emissions from reservoir at O&M stage

a: Civil work

b: Electronic equipment

√: GWP is reported as total life cycle GHG emissions while emissions from each life stage are not provided

Table D2. Estimated reservoir GHG emissions from the three pathways by the previous studies

Dam /reservoir	Location	Hydropower capacity (10^8kWh/yr)	Reservoir surface emission			Degassing	Downstream emission (Mt/yr)		Annual emissions (Mt/yr)		Total emissions (g CO_2 eq./kWh)		Ref.
			Diffusion (Mt/yr)		Ebullition (Mt CH_4/yr)								
			CO_2	CH_4			CO_2	CH_4	CO_2	CH_4	20-yr GWP[a]	100-yr GWP[a]	
Boreal													
Sainte-Marguerite	Canada	27.7	0.02	0					0.02	0	7	7	(McCully, 2006)
Churchill /Nelson	Canada	140	0.22	0.003					0.22	0.003	34	23	(McCully, 2006)
Manic Complex	Canada	200	0.64	0.008					0.64	0.008	66	46	(McCully, 2006)
La Grande Complex	Canada	820	3.28	0.039					3.28	0.039	81	56	(McCully, 2006)
Churchill Falls	Canada	350	1.67	0.020					1.67	0.020	97	67	(McCully, 2006)
Temperate													
Hydro-reservoir	China	6540	29.6	0.47					29.6	0.47	107	70	(Li, S. et al., 2015)
Segredo	Brazil	55.2	N/A	N/A	N/A				0.08	0.0003	19	16	(Demarty and Bastien, 2011)
Itaipu	Brazil	900	N/A	N/A	N/A				0.05	0.0059	6	3	(Demarty and Bastien, 2011)
Tropical													
Curua-Una	Brazil	1.9	0.04	0.023							10621	4326	(McCully, 2006)
Tucurui	Brazil	180	9.34	1.064							5602	2529	(Li, S. et al., 2015)
Samuel	Brazil	5.3	0.65	0.007	0.0008		0.021	0.0004	0.86	0.008	2984	2170	(Guérin et al., 2006)
Balbina	Brazil	9.7	1.96	0.017	0.005		0.16	0.007	2.11	0.029	4731	3186	(Guérin et al., 2006)
Petit Saut	French (Guyana)	4.7	0.53	0.002	0.0004		0.13	0.001	0.67	0.003	2105	1680	(Guérin et al., 2006)

a: CH_4 20-yr GWP = 86; 100-yr GWP = 34

Table D3. Range of GHG emissions (g CO_2 eq./ kWh) of different hydropower projects (HPs)

Type	Construction	O&M	End-of-life		Total (LCA study)	Reservoir emission (Not LCA study)
			Dam removal	Sediment release		
River in-stream HPs	N/A	N/A	N/A	-	2.1-4.8	-
Diversion HPs	2.3-37.9	0.9-37	0-0.1	-	5.5-74.9	-
Reservoir-based HPs	2.3-33	3.7-77	-	40-148	6.1-227.3	7-67 (Boreal)
Pumped-storage HPs	4-5	1.8-640	1	-	4.8-967.5	3-70 (Temperate) 8-6647 (Tropical)

APPENDIX E: SUPPORTING INFORMATION FOR CHAPTER 6

Section E1: Estimation of river flow in the Pearl River basin

The drainage The bankfull channel geometry (e.g., width, mean depth, and cross-sectional area) and discharge were estimated based on the following regression equations provided in (Bent and Waite, 2013).

Bankfull width (ft) = 15.0418 × (drainage area (mi^2))$^{0.4038}$

Bankfull mean depth (ft) = 0.9502 × (drainage area (mi^2))$^{0.2960}$

Bankfull cross-section area (ft^2) = 14.1156 × (drainage area (mi^2))$^{0.7026}$

Bankfull discharge (ft^3/s) = 37.1364 × (drainage area (mi^2))$^{0.7996}$

The bankfull discharge is the discharge that fills a stable alluvial channel up to the elevation of the active floodplain. In many natural channels, this is the discharge that just fills the cross section without overtopping the banks, hence the term 'bankfull'.

Table E1. Baseline stream data (in standard metric units).

Dams	Drainage area at each dam site (km^2)	Calculated bankfull width (m)	Calculated bankfull mean depth (m)	Calculated cross-section area (m^2)	Calculated bankfull discharge (m^3/s)	Calculated bankfull discharge (m^3/d)
Dam 1	466	37	1.3	50	67	5,789,000
Dam 2	389	35	1.3	44	58	5,011,000
Dam 3	130	22	0.9	20	24	2,074,000
Dam A	181	26	1	26	31	2,678,000
Dam B	104	20	0.9	18	20	1,728,000

Section E2: Preferred alternative for each stakeholder

According to the primary interests of each designed role, each role's preferred dam management alternative was identified and listed in Table 6-6. These preferred dam management alternatives were listed in Diessner et al., 2020. For simplification and illustration, we only analyzed one preferred dam management alternative for each role. It should be noted that each role may have more than one preferred dam management alternatives because similar outcomes could be achieved through different basin-scale dam management options. Additionally, the preferred dam management alternative could vary from player-to-player, and as long as the player follows the constraints of their assigned role, it is possible to have numerous "preferred" alternatives.

Table E2. A preferred dam management scenario of each role based upon their primary interests

Role	Dam management alternatives					Normalized values of six performance indicators					
	Dam-1	Dam-2	Dam-3	Dam-A	Dam-B	Energy	Cost	Alewife	Shad	Salmon	Sea lamprey
FANR	Remove	Install Denil	No action	Remove	Remove	0.44	0.47	0.88	0.86	0.99	0.79
WRD	Remove	No action	No action	Remove	Remove	0.44	0.59	0.75	0.72	0.98	0.45

HPAS, Town	No action	No action	No action	Repair	No action	0.84	0.96	0	0	0	0
Hydro Energy LLC., HOA	No action	No action	No action	Repair, install turbine and nature-like fishway	No action	0.94	0.84	0	0	0.37	0
Rivers-R-Us	Remove	Remove	Remove	Remove	Remove	0	0.25	1	1	1	1

Under these preferred dam management alternatives, the normalized values of six performance indicators are provided in both Table 6-6 and Figure 6-4 (A). The results shown that the studied seven roles can be roughly divided into three groups based upon their preference for energy, fish, and cost. One group includes four roles: HPAS, Town, HydroEnergy LLC, and HOA. These four roles mainly pay their attention on historical values, property values, pond-based recreation, and safety issues related to Dam-A. Therefore, their preferred management actions are solely about managing (e.g., repair, install turbine and fishway) Dam-A. Although these actions have superior performance in energy generation and project cost, fish population potential of four fish species is relatively low. Another group involves FANR and WRD. For these two roles, they mainly focused on improving fish populations as well as ecosystem health and resilience. Their preferred management actions listed here is removing the most downstream dam and the two tributary dams which lead to significant increase in fish population along with losing around half of energy generation and costing around 50% of the maximum cost. The last group is River-R-Us who cares about not only improving fish populations and ecosystem health but also enhancing river-based recreation. One preferred management alternative for this role is removing all five dams where extreme conflicts between fish and energy happen in accompany with relatively high project cost.

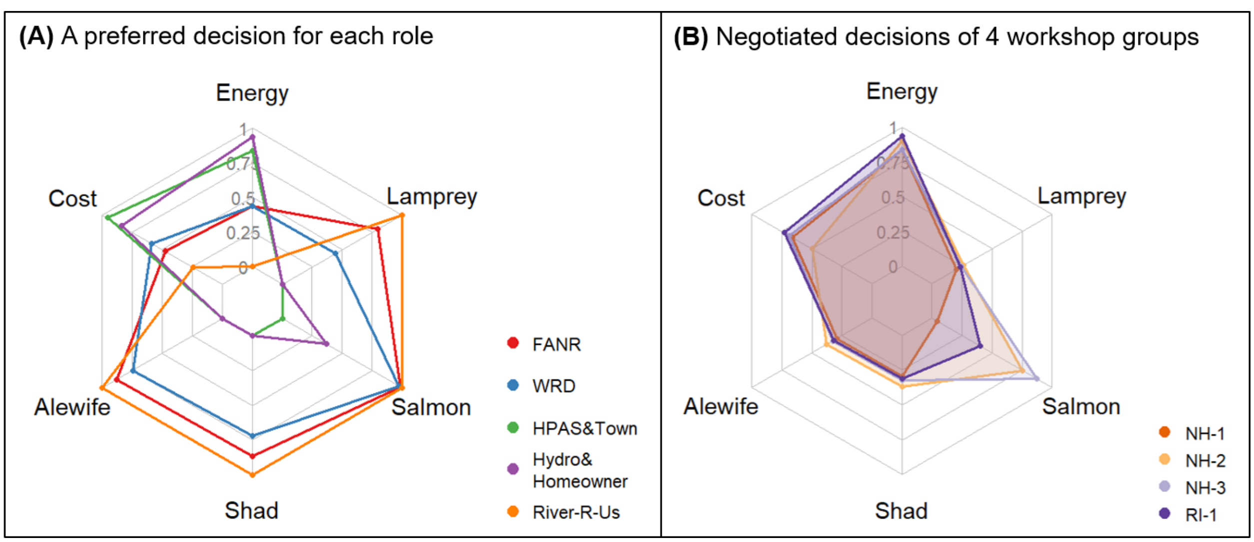


Figure E1. Radar chart presenting performances of energy, cost, and populations of four fish species under (A) preferred dam management scenarios for all seven roles, (B) negotiated decisions at four workshop groups.

For all four negotiated decisions, the extent of gain/loss each stakeholder achieved based upon their preferred alternatives is provided in Table 6-7. Under the first weighting scenario, compromises were always made by participants playing the roles of River-R-Us, FANR, and WRD at all four negotiating groups because they agreed on negotiated agreements that did not meet their most preferred outcome for populations of each of the four fish species. The extent of loss among these three roles is in the following sequence: FANR > River-R-Us > WRD. For the remaining four roles (HPAS, Town, HydroEnergy, and HOA), participants playing these roles gains at NH-2, NH-3, and RI-1 workshop groups. The extent of gain achieved by HPAS and Town is larger than HydroEnergy, and HOA. Extremely different results of each stakeholder's gain and loss were obtained when applying the second weighting scenario. River-R-Us is the only role who gains at all four workshop groups. At the NH-1 workshop group, losses come to all roles except for River-R-Us with the following sequence according to extent of loss: HydroEnergy and HOA > FANR, HPAS, and Town > WRD. At the NH-2 workshop group, HydroEnergy and HOA are the two roles who loss. Under the NH-3 and RI-1 negotiated decisions, gains achieved by all seven roles and the extent of gains is as follows: River-R-Us > WRD > FANR, HPAS, and Town > HydroEnergy, and HOA.

Table E3. Extent of gain/loss of each role under four negotiated decisions based on a preferred alternative of each stakeholder

Role	Extent of gain/loss (w_i = 1 for all six system indicators)				Extent of gain/loss (w_i = 1 for energy and cost indicators, w_i = 0.25 for fish indicators)			
	NH-1	NH-2	NH-3	RI-1	NH-1	NH-2	NH-3	RI-1
FANR	-2.11	-1.50	-1.15	-1.50	-0.09	0.19	0.17	0.19
WRD	-1.59	-0.75	-0.63	-0.98	-0.04	0.10	0.22	0.23
HPAS and Town	0.54	1.37	1.49	1.14	-0.09	0.05	0.17	0.19
HydroEnergy, LLC. and HOA	0.19	1.02	1.14	0.79	-0.16	-0.03	0.09	0.11
Rivers-R-Us	-1.91	-1.08	-0.96	-1.31	0.46	0.60	0.72	0.74

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