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A temporal perspective to dam management: Influence of dam life and threshold fishery conditions on the energy-fish tradeoff

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Abstract

While hydroelectric dams play a significant role in meeting the increasing energy demand worldwide, they pose a significant risk to riverine biodiversity and food security for millions of people that mainly depend upon floodplain fisheries. Dam structures could affect fish populations both directly and indirectly through loss of accessible spawning and rearing habitat, degradation of habitat quality (e.g., changes in temperature and discharge), and/or turbine injuries. However, our understandings of the impacts of dam life span and the initial fishery conditions on restoration time and hence the dynamic hydropower (energy)-fish (food) nexus remain limited. In this study, we explored the temporal energy-food tradeoffs associated with a hydroelectric dam located in the Penobscot River basin of the United States. We investigated the influence of dam life span, upstream passage rate, and downstream habitat area on the energy-food tradeoffs using a system dynamics model. Our results show that around 90% of fish biomass loss happen within five years of dam construction. Thereafter, fish decline slowly stabilizes and approaches the lowest value at around 20th year after dam construction. Fish restoration period is highly sensitive even to a short period of blockage. The biomass of alewife spawners need 18 years to recover with only one-year of blockage to the upstream critical habitats. Hydropower generation and loss of fish biomass present a two-segment linear relationship under changes in dam life span. When the dam life span is less than five years, generating 1 GWh energy cause around 0.04 million kg loss of fish biomass; otherwise, the loss of fish biomass is 0.02 million kg. The loss of fish biomass could be significantly decreased with minimal energy loss through increasing upstream passage rate and/or the size of downstream habitat area.

Keywords:

System dynamics modeling, energy-fish nexus, hydroelectric dam construction and removal, diadromous fish restoration.

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; Ziv et al. 2012; Winemiller et al. 2016; Wild et al. 2018). There are more than 45,000 large dams (>15 meters in height) around the world, which are mainly used for hydropower generation and irrigation (WCD 2000; Vörösmarty et al. 2010; Bartle 2002). 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 (Zarfl et al. 2015; Winemiller et al. 2016; Limburg and Waldman 2009; Song et al. 2018; Chen et al. 2016). Dams can substantially decrease fish populations by fragmenting migration corridors (Hall et al. 2011; Beasley and Hightower 2000), 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 (ICEM 2009; Grumbine and Xu 2011). 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; Song et al. 2019; Roy et al. 2018; Ziv et al. 2012). Other studies have also found that installing effective fish upstream passage structures (hereafter referred to as fishways) (Null et al. 2014; Thorncraft and Harris 2000; Katopodis and Williams 2012; Larinier 2000) and properly shutting down turbines during fish peak downstream migration period (Trancart et al. 2013; Watene and Boubée 2005; Eyler et al. 2016) 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.

2. Materials and Methods

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 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. 2013; Gardner et al. 2012; Gehrke 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 (Dalton et al. 2009; McClenachan et al. 2015; ASMFC 2009). 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 (MaineDMR 2018; Goode 2006). This dramatic decline was considered to be attributed to existing dams.

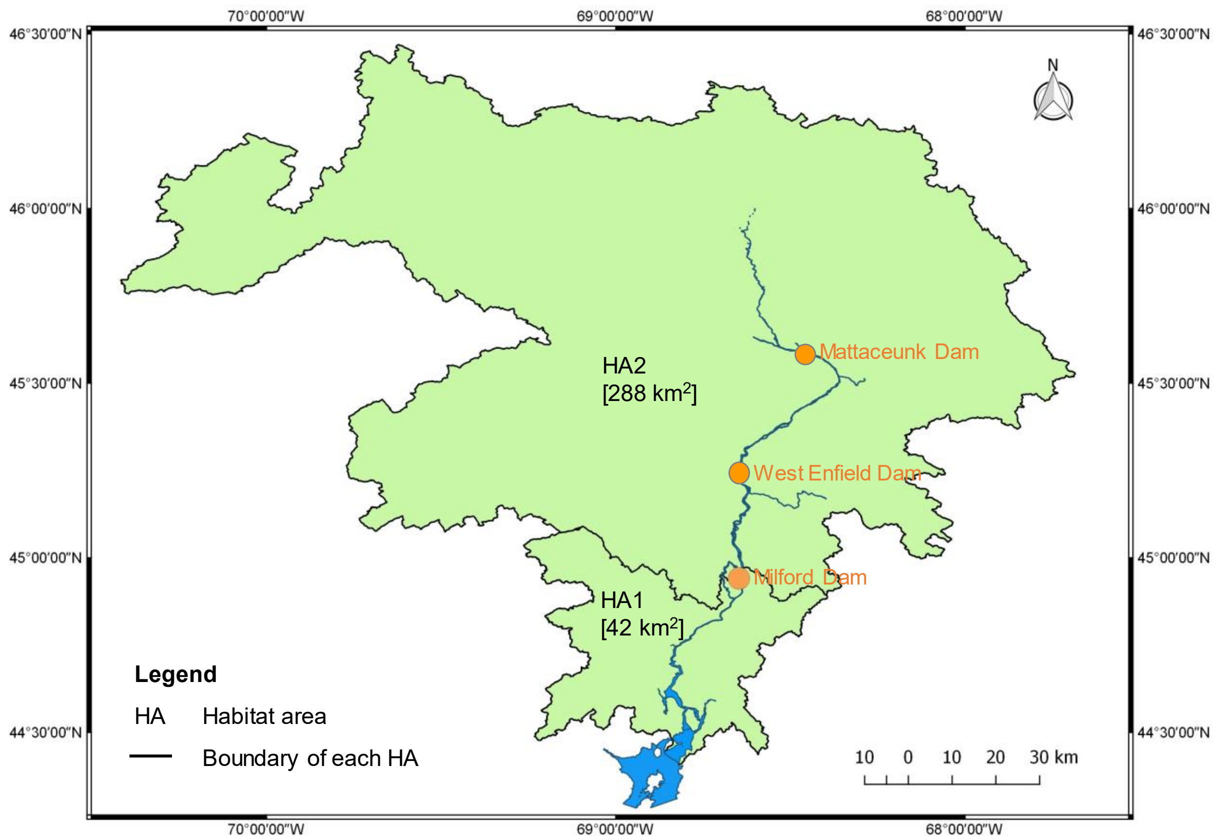


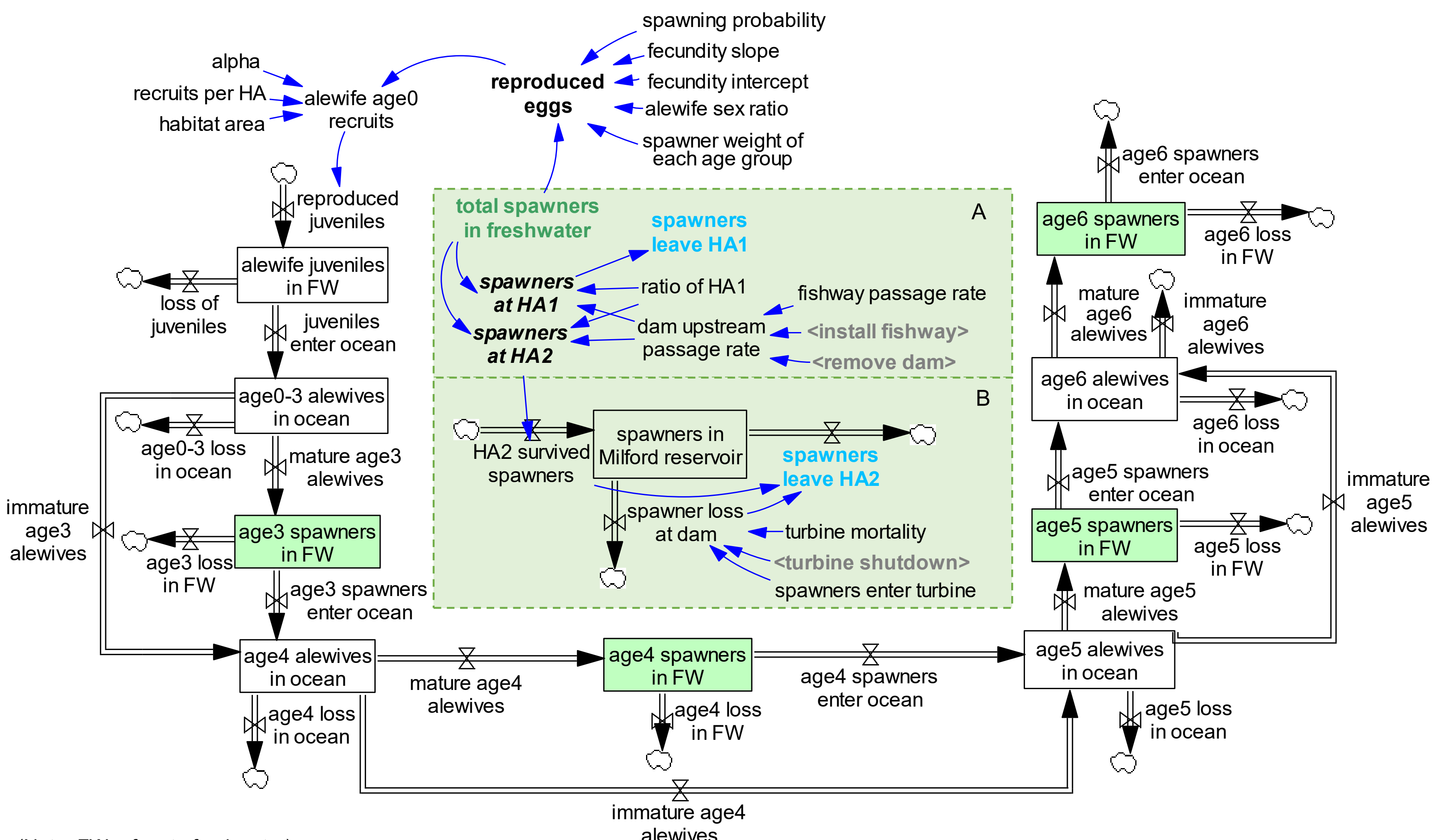
Figure 1. Map of the study area showing the locations of Milford Dam as well as the size of alewife spawning lakes/ponds in the Penobscot River Basin.

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 2018), 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 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 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 supporting information. 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 the supporting information.



(Note: FW refers to freshwater)

Figure 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 supporting information. 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 3 shows an abstracted version of the stock-and-flow diagram of the hydropower generation model (see supporting information 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 (Singh and Singal 2017; Hadjerioua et al. 2012; Power 2015; Adeva Bustos et al. 2017).

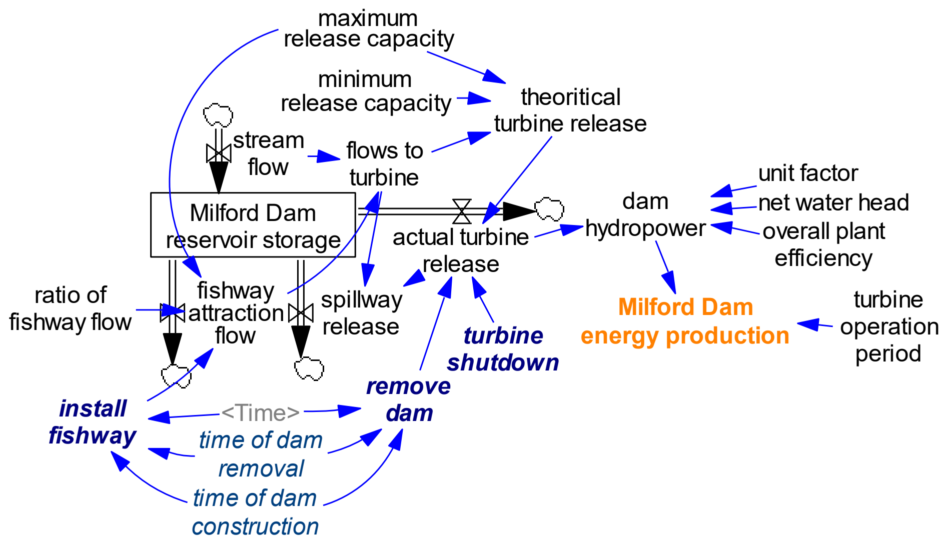


Figure 3. Schematic stock-and-flow diagram of energy generation model.

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.

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.

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 (Sterman 1984; Ventana 2002; Cheng et al. 2018). All constant variables within the base model were varied by -20% to 20% of their original values as provided in Table 1. The Monte Carlo simulation was repeated for 200 times.

Table 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]

3. Results and Discussions

3.1 Energy-fish nexus under different dam life spans

Figure 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 (Song et al. 2019; Maynard et al. 2017). 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; Ford 2000; Einum and Fleming 2000; Mousseau and Fox 1998).

Dam life span determines the number of initial fishery populations at the time of dam removal (Figure 4 (a)), as well as the time needed to restore alewife population to its pre-damming level (Figure 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 (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.

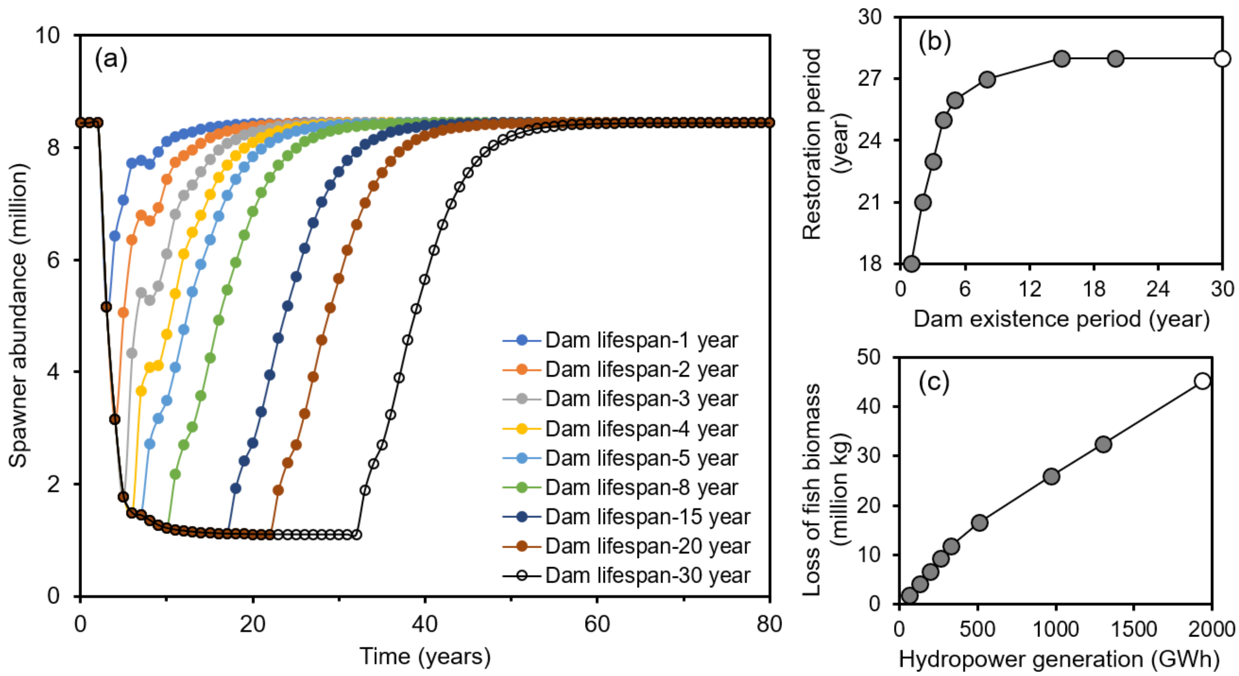


Figure 4. Temporal changes of alewife spawner abundance with dam life span (a), relationship between dam life span and fish restoration period (b), and the energy-fish nexus (c).

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 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 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 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 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.

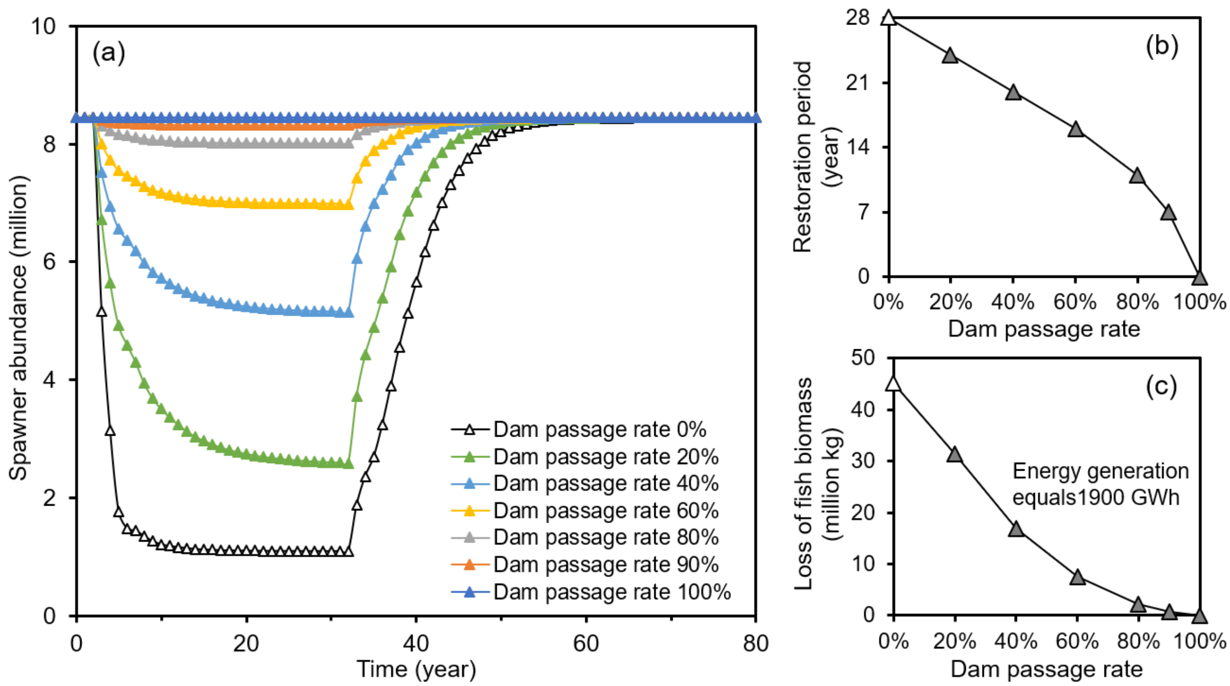


Figure 5. Temporal changes of alewife spawner abundance with upstream passage rate (a), relationship between dam passage rate and fish restoration period (b), relationship between dam passage rate and loss of fish biomass (c).

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 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 6 (a)), and a longer fish restoration period (Figure 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 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 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.

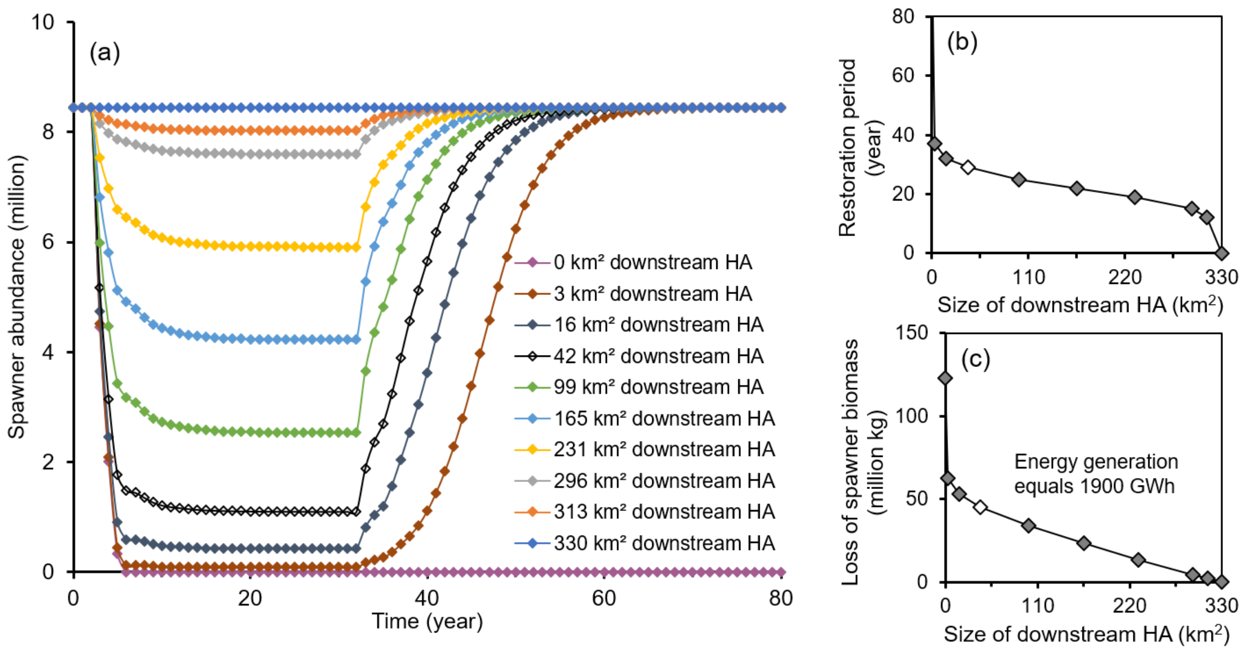


Figure 6. Time-series changes of alewife spawner abundance with size of downstream habitat area (HA) (a), relationship between size of downstream habitat area and fish restoration period (b), relationship between size of downstream habitat areas and loss of fish biomass (c).

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 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 Deriso 1999; Quinn and Collie 2005).

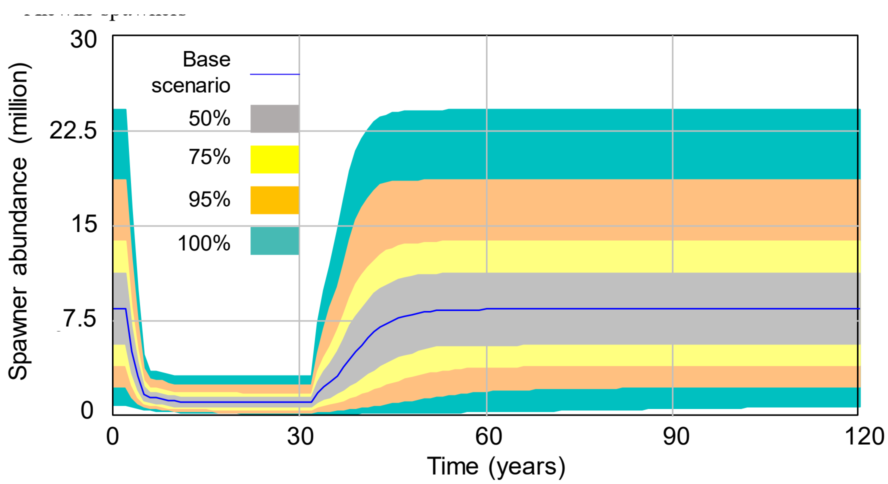


Figure 7. Monte Carlo simulation of the age-structured fish population model

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).

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