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# Modeling Channel Response to Dam Removal in Lansing, Michigan, Using SWAT

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MODELING CHANNEL RESPONSE TO DAM REMOVAL IN LANSING, MICHIGAN, USING SWAT

by

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The removal of dams has increased in recent decades in the United States, largely resulting from decaying infrastructure and greater efforts to restore rivers to a more natural, free-flowing state. Dam removal presents the opportunity for increased public safety, improved environmental prosperity, and improved economic prosperity in conjunction with riverfront revitalization projects. The City of Lansing, Michigan, contains two moderate-to high-risk dams along the Grand River that pose a significant risk to the surrounding area in the event of structural failure.

The Soil and Water Assessment Tool (SWAT) is applied to model the impacts of the Moores Park Dam and the North Lansing Dam on streamflow magnitude within downtown Lansing. The study used SWAT to recreate conditions in the Grand River watershed to approximate the differences in stream discharge with the dams in place and with the dams removed. It was hypothesized that removal of these structures will coincide with a decrease in stream discharge and downstream flooding concerns. Despite adjusting hydrologic parameters that effect the watershed, the model was unable to replicate baseline watershed conditions. Future research could be improved with more primary data collected in field studies.

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CHAPTER I

INTRODUCTION

**Background**

With a length of 260 miles and a watershed drainage area of 5,572 $mi^2$, the Grand River is the longest river in Michigan, and the watershed is the second largest in the state. The river flows through and drains portions of 15 counties from its headwaters near Jackson to its terminus in Grand Haven. Land use within the watershed is mixed between agricultural, forest, wetland, and urban settings including the cities of Jackson, Lansing, Grand Rapids, and Grand Haven (The Grand River Watershed – Michigan, n.d.). The land use for the watershed is shown in Figure 1, using the United States Geological Survey (USGS) Land Cover Institute's (LCI) National Land Cover Database (NLCD) for 2011.

Several dams along the Grand River have a high hazard potential because major structural failure would result in major property damage and/or the loss of life (Hanshue & Harrington, 2011). The Moores Park and North Lansing Dams are deteriorating structures presenting a threat to the downtown Lansing area. These dams are identified as significant risks because of the potential impacts if they were to fail, not necessarily because of the respective structural integrities (Dam Failure, n.d.). The relative locations of these dams in respect to the Grand River Watershed are shown in Figure 2, with dam locations derived from the U.S. Army Corps of Engineers (USACE) National Dam Inventory.

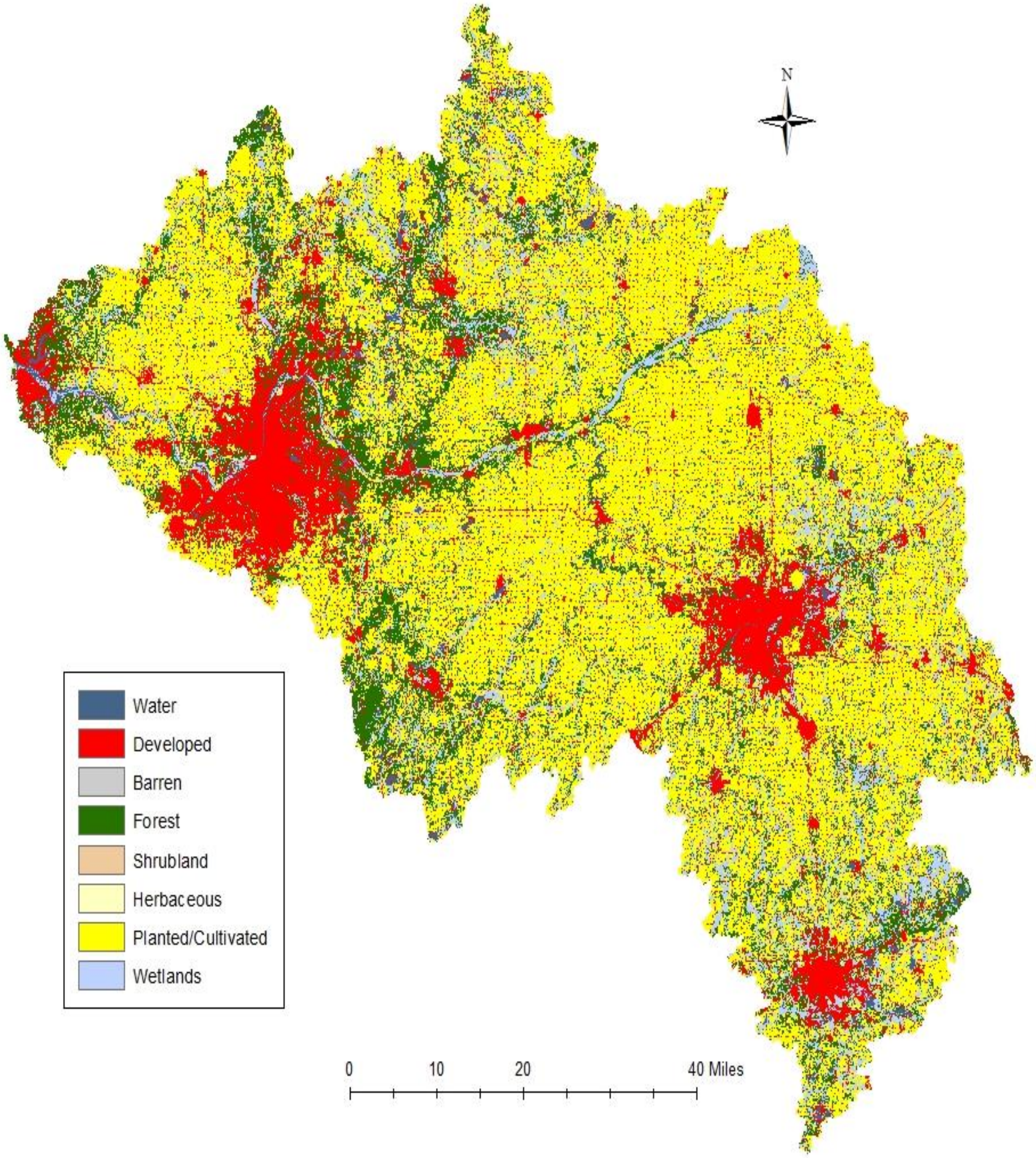


*Figure 1: Land Use in the Grand River Watershed, 2011*

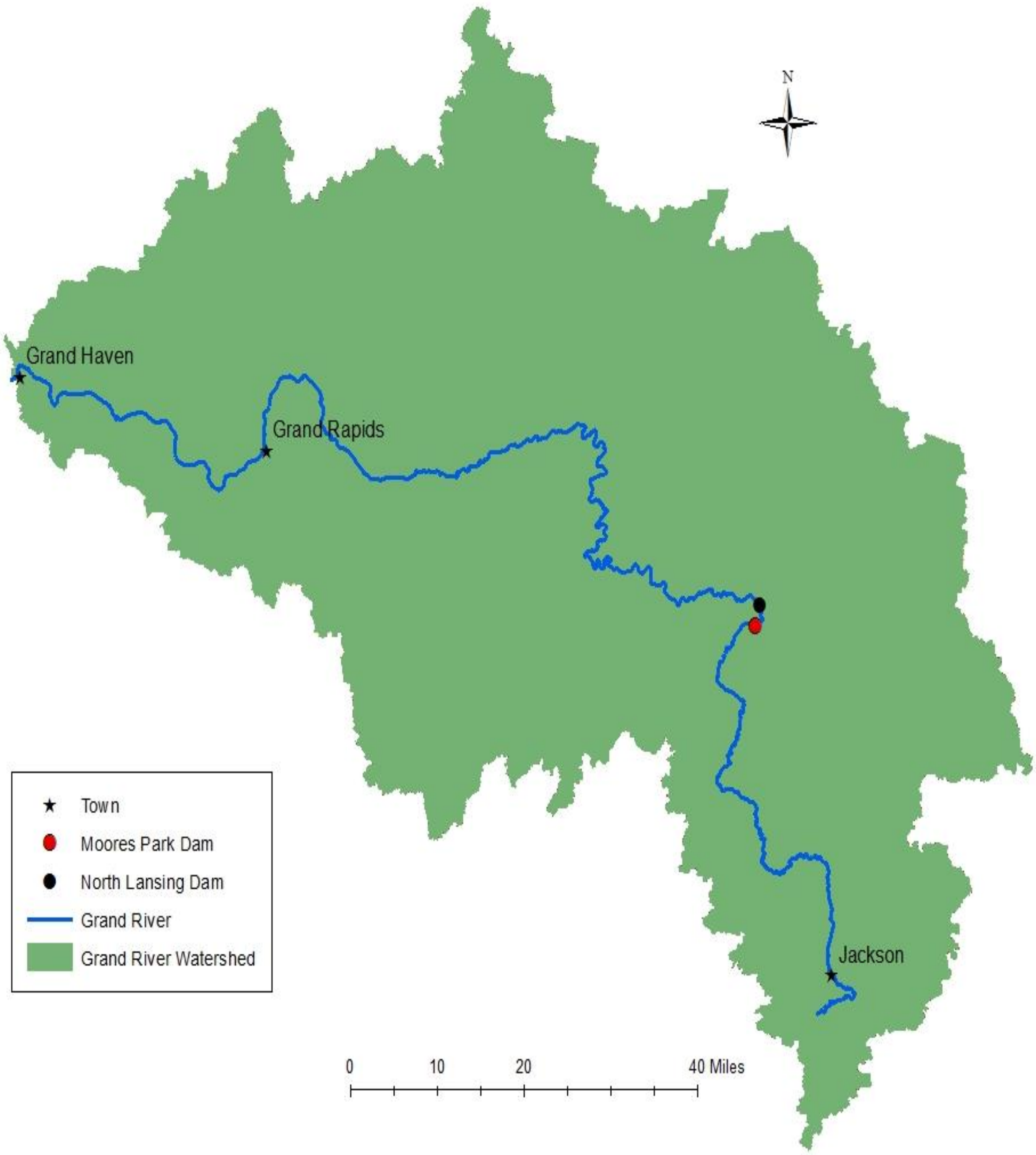


*Figure 2: Location of Dams of Concern in the Grand River Watershed*

**Statement of Purpose**

Lansing city officials have identified dam removal as the preferred mitigation technique for reducing the risk of dam failure. Any efforts to remove the Moores Park Dam and the North Lansing Dam necessitate considerable caution given the dense population of the surrounding area and the potential for significant fluvial impacts. Project planning must account for changes to the streamflow behavior following dam removal.

This research uses the Soil and Water Assessment Tool (SWAT) to simulate stream flow on a current time scale without the presence of either dam along the Grand River. Calibration and validation of a model for the Grand River Watershed under a dam-in scenario allows comparisons of simulated data to observed, secondary data from the USGS. Calibration and validation under a dam-out scenario is unattainable because of the incomplete data record on file with the USGS from the early 1900s. A sensitivity analysis preceded model calibration and validation, allowing for determination of the most significant input parameters affecting model output, as described by Arnold et al. (2012a).

The purposes of this research are:

1) To effectively model baseline conditions in the Grand River Watershed;
2) To determine the difference in streamflow magnitude between baseline conditions and a “dam-out” scenario;
3) To relate modeling results to potential mitigation and management scenarios for the dams and surrounding area

The null hypothesis for this study was no significant change in streamflow magnitude and flooding risk with the dams in place versus the dams not in place. Based on a Draft Grand River Assessment by Hanshue & Harrington (2011), I expected a decrease in stream flow magnitude and a decrease in flooding risks. Impoundment removal will improve downstream transport of large woody debris (LWD) during higher flow events. Furthermore, dam removal will lower baseline water levels behind the structure and increase channel carrying capacity.

**Project Scope**

This study utilized SWAT to simulate streamflow conditions in the Grand River in downtown Lansing under the baseline scenario and under the dam removal scenario. Successful simulations will help determine the degree to which the Moores Park Dam and the North Lansing Dam affect streamflow within the study area.

Methods and results of this research may be shared with city officials in Lansing, and with organizations in the City of Grand Rapids, where the removal of downtown dams along the Grand River has been proposed. Modeling results for Lansing may provide a template for future modeling studies of dam removal in Grand Rapids, or any area exploring dam removal options.

The rest of this thesis will review literature related to the impacts of dams, the restoration of rivers from removing dams, the history of dams in the Grand River watershed, and modeling techniques implemented to analyze the impact of dams on watersheds. The discussion of modeling techniques will further describe SWAT and the model layout. The methodology also describes the layout and functionality of SWAT, along with the methodological approach of this research. Results of the simulations and their significance are then discussed. The overall successes and limitations of the research are outlined.

# CHAPTER II

# LITERATURE REVIEW

## Background

River regulation and dam construction in the United States expanded greatly in the early to mid-1900s, coinciding with technological advances and industrial pursuits. Increased dam infrastructure provided regional benefits for crop irrigation, public water supply, and hydroelectric power (Graf, 1999). Dam construction intensified between 1935 and 1965, with notable constructions of Hoover and Glen Canyon Dams in the southwest U.S. (Mertha & Lowry, 2006), and the formation of the 26-dam Tennessee Valley Authority system (TVA – our history, n.d.).

Much of the older dam infrastructure in the United States continues to crumble, spurning an increase in dam removal beginning in the 1990s through today. In the U.S., the average life expectancy of a dam is roughly 50 years (Mission 2012: clean water, n.d.). The geomorphic response to dam removal is not widely understood and is conditional to each river system. Each system has a different morphology related to the construction and presence of a dam during its operation.

As dam removal and river restoration projects have increased, public understanding of the overall impacts of dams has also expanded. In 1999, former U.S. Secretary of the Interior, Bruce Babbitt, spoke to the Ecological Society of America regarding the fragile state of dams in the U.S. Many of the 75,000 dams constructed in the U.S. have outlived their function while also continuing to play a role in the destruction of ecological habitats along river systems. The Colorado River represents a system significantly harmed by damming, as the once mighty river no longer reaches the ocean (Babbitt, 1999). The Glen Canyon Dam's Lake Powell and the Hoover Dam's Lake Mead collectively account for much of the water storage in this river system which provides water to one in eight Americans. However, less water flows into each reservoir than is taken out, as increasingly dry conditions coincide with increasing water demand and overuse (Lustgarten, 2016).

The US Army Corps of Engineers lists 941 Michigan dams in its National Inventory Report. Of these 941 dams, 322 are classified as having a moderate-to high-risk potential. Much like the Moores Park and North Lansing Dams, this risk assessment is more related to the potential effect on communities in the event of a structural failure, not necessarily the imminent risk of a failure. Furthermore, more than half of the dams in Michigan (519) were constructed prior to 1960.

Of the 941 dams in Michigan registered in the National Inventory Report, 755 of these are smaller structures under 25ft. in height. The State of Michigan does not disclose a classification scheme for determining if a dam is a large dam, intermediate dam, small dam, or minor dam. However, the neighboring state of Ohio classifies dams based on size and hazard parameters. Small dams in Ohio belong to Class III, defined as having a total storage volume exceeding 50 acre-feet, or a height exceeding 25 feet (State Dam Safety Dam Size Classification Schemes, 2010).

**Impacts of Dams**

Dam construction in the last 100 years, in conjunction with global industrialization, has been vital to increased economic productivity and viability for many regions. The geologic setting of the area, mechanisms and magnitude of sediment transport, channel processes, and disturbances drive the response of a river and watershed to impoundment (Grant et al., 2003). Dams have a significant impact on the adjacent ecological habitat and landscape. These structures serve as barriers that influence downstream streamflow conditions and sediment transport, causing a change in thermal regimes and the function of riparian and aquatic habitats (Poff & Hart, 2002).

Finer-grained sediments are more likely to be transported downstream, while coarser sediments are likely to become stored and trapped behind the impoundment. Sediment transport is impacted immediately downstream of the dam and potentially throughout the remainder of the watershed. For instance, the Mississippi River has observed a decreased by one-half in suspended sediment transport to the Gulf of Mexico since the early 1700s (Meade, 1995).

Changes in the downstream thermal regime are advantageous for cold-blooded species and detrimental to warm-blooded species (Ward & Stanford, 2013). In Wisconsin, freshwater systems with impoundments have a higher likelihood for non-native invasive species than natural freshwater systems (Johnson et al., 2008). Wide-spread damming along a river and within individual watersheds can have the cumulative effect of fragmenting the ecosystem, as seen with salmon stocks in the northwest US. (Bjornn, 1998).

While there is a continued shift from dam construction to dam removal, the impacts of removal on stream ecology and stream geomorphology is not well understood (Grant, 2001). Although each river system responds uniquely to dam removal, expanded model development would benefit future environmental impact studies and subsequent management practices. The stress of climate change on the world's river systems, which are already under a great deal of stress from land use, urban development, and impoundments, must be considered.

Significant environmental impacts are probable from ongoing neglect of a deteriorating dam. A study by Evans et al. (2000) discussed the increased need for dam removal research through analyzing the Upper Mill Pond (IVEX) Dam failure on the Chagrin River in Ohio in 1994. A 70-year rainfall event caused significant flow over a spillway, impinging the top of the dam. While the historic rainfall contributed to structural failure, the main cause was excessive buildup of sediment (236,000 $m^3$) behind the dam, which lowered water storage capacity by 86 percent. (Evans et al., 2000).

Dams and dam removal also affect real estate economics. Assessing the political and economic repercussions of dam removal, Lewis et al. (2008) studied the effect of dams on property values along the Kennebec River in Maine, where one hydroelectric dam (Edwards Dam) was removed in 1999, and two other hydroelectric dams remain in use. Using real estate information and a hedonic property valuation method, which assesses marginal prices of different attributes of housing choice, the willingness to pay to be farther from a dam site dramatically decreased after removal of the Edwards Dam.

**River Restoration via Dam Removal**

Much of the motivation for dam removal is to restore a river to a more natural, free-flowing state. Removing failing structures may help increase habitat biodiversity, improve water quality, and improve overall ecosystem health in the affected areas upstream and downstream of the site.

Although a dam disrupts the preexisting natural processes within a river system and ecosystem, there is an adjustment to a new equilibrium over time. Recall that the Moores Park Dam in Lansing was erected in 1904, meaning that this impoundment has affected the nearby area in the Grand River for over 100 years. The longevity of this structure has allowed the surrounding environment to adjust and reach a new baseline for thermal regimes, streamflow, sediment transport and fluxes, and nutrient loads.

The Woolen Mills Dam along the Milwaukee River in Wisconsin underwent removal to improve aquatic biodiversity. Retiring this structure also sought to increase the population of native fish species such as smallmouth bass, and to decrease the population of invasive fish species such as the common carp, which prefers the gentler river conditions associated with reaches regulated by dams. Kanehl et al. (1997) developed a habitat index to assess the strength of a habitat along the Milwaukee River following impoundment removal. An index score closer to 0 indicates a more unsuitable habitat, while a score closer to 99 indicates a more healthy and viable habitat for native species. Their study determined that locations along the river near the dam site had the most significant habitat index improvement. For the Milwaukee 2 field station, one of two former impoundment stations in the study area, the habitat index increased from 24 prior to dam removal in 1988 to 60 in 1993 (Kanehl et al., 1997).

Dam removal presents the opportunity for improved ecological health, and the opportunity to disrupt the baseline equilibrium in place. The removal of the Dead Lake Dam (Florida), Edwards Dam (Maine), and Elwha Dam (Washington) yielded improved spawning grounds for fish, improved fish passage, improved sediment transport, and improved water quality (Bednarek, 2001). However, other removal efforts, such with the Fort Edwards Dam (New York) and Fulton Dam (Wisconsin), have negatively affected ecology through changes in the thermal regimes, changes in community composition within the ecosystem, the loss of reservoir species, and the release of toxic polychlorinated biphenyls (PCBs) into downstream locations from the dam site (Bednarek, 2001).

Table 1 is a summary of completed and proposed dam removals across the U.S., and the overall significant ecological impacts. Data in Table 1 reflects available data at the time of the study. The Elwha Dams and Stronach Dam were not removed at the time of the study, but have subsequently been removed.

*Table 1: Significant Ecological Impacts of Dam Removal*

| Dam | Location | Removal Date | Ecological Impacts | Reference |
|---|---|---|---|---|
| Dead Lake Dam | Chipola River, Florida | Dec. 1987 | Improved fish passage and water quality; greater fish species diversity | Hill and others 1993, Estes and others 1993 |
| Edwards Dam | Kennebec River, Maine | July 1999 | Sediment changes; improved fish passage | Dadswell 1996 |
| Elwha Dams | Elwha River, Washington | Not yet removed | Native species return; improved coastal sediment transport | DOI 1995 |
| Enloe Dam | Similkameen River, Oregon | Not yet removed | Improved fish passage | Winter 1990 |
| Fort Edwards Dam | Hudson River, New York | Breached in 1973 | Released PCBs | Shuman 1995, Chatterjee 1997 |
| Fulton Dam | Yahara River, Wisconsin | 1993 | Change in community composition; loss of reservoir species | ASCE 1997, Born and others 1998 |
| Grangeville Dam | Clearwater River, Idaho | 1963 | Improved sediment movement | Winter 1990 |
| Lewiston Dam | Clearwater River, Idaho | 1973 | Improved sediment transport | Winter 1990 |
| Little Goose Dam | Snake River | Not yet removed | Improve fish passage | Wik 1995 |
| Newaygo Dam | Muskegon River, Michigan | 1969 | Sediment release | Simons and Simons 1991 |
| Rodman Dam | Oklawaha River, Florida | Not yet removed | Improved mammal and waterfowl habitat | Kaufman 1992, Shuman 1995 |
| Sallings Dam | AuSable River, Michigan | 1991 | Temperature changes | Pawloski and Cook, 1993 |
| Stronach Dam | Pine River, Michigan | Undergoing removal | Improved sediment and fish movement | American Rivers and others 1999 |
| Sweasey Dam | Mad River, California | 1969 | Reservoir silted in; improved fish passage | Winter 1990 |
| Woolen Mills Dam | Milwaukee River, Wisconsin | May 1988 | Sediment release; improved organism movement | Nelson and Pajak 1990, Staggs and others 1995, Kanehl and others 1997 |
| Washington Water Power Dam | Clearwater River, Idaho | 1963 | Improved fish passage and chinook salmon habitat | Shuman 1995 |

*Source:* Bednarek, 2001.

Efforts to restore a natural river habitat have not always proven successful. For instance, the Boardman River drains its 287-$mi^2$ watershed near Traverse City into West Grand Traverse Bay in the northwest Lower Peninsula of Michigan (The Boardman: A River Reborn, n.d.). As part of recommendations from the Michigan Department of Environmental Quality (MDEQ), the City of Traverse City and Grand Traverse County launched the Boardman River Dams Ecosystem Restoration Project to remove three dams and repurpose another (Thompson, 2015). As part of this project, the Brown Bridge Dam was first scheduled for removal. The dam was constructed in 1921 for providing hydroelectric power to Traverse City (The Boardman: A River Reborn, n.d.). Dam construction resulted in the formation of a 170-acre pond upstream of the dam (Thompson, 2015).

Removal of the Brown Bridge Dam began in 2011, with restoration of the original river channel, the excavation of 250,000 $yd^3$ of pond sediment, and the construction of a drawdown structure to gradually release pond water included in the project scope. The drawdown structure was compromised in October 2012, when the sandy soils beneath the structure became saturated and flushed out the structure. Flash-flooding and sediment deposition resulted from the structural collapse, causing significant damage to 66 downstream residential properties. Affected homeowners filed a lawsuit against the City of Traverse City and Molan Excavating, Inc., the firm contracted to remove the dam. The lawsuit settlement was finalized in December 2014. The terms of the settlement have not been made public (McGillivary, 2014). Additionally, the ecological and fluvial impacts of the dam removal are yet to be quantified.

**Dam Management in the Grand River Watershed**

The presence of dams significantly impacts the Grand River Watershed. There are 231 registered dams within the watershed. Including the Moores Park Dam and North Lansing Dam, 30 percent of dams were constructed prior to 1960 and have outlived their function ability (Hanshue & Harrington, 2011). Many of these structures along the Grand River and elsewhere in the watershed are low-head dams. Low-head dams, or run-of-the-river dams, are typically 3 to 5 meters in height, with streamflow over the entire structure to raise water levels for industrial and/or recreational purposes (Tschantz & Wright, 2011). In many instances, the US Army Corps of Engineers' National Dam Inventory does not have a data record for storage capacity, structure width and height, and other features for smaller low-head impoundments, such as the Sixth Street Dam and two other small dams in Grand Rapids.

The US Army Corps of Engineers classifies high-risk dams as being hazard type 1 (dam failure resulting in the loss of life), hazard type 2 (dam failure resulting in severe property damage), and hazard type 3 (a low-head dam in a remote area). Of the 231 registered structures in the Grand River watershed, 27 are classified as high-risk structures, with eight of these high-risk structures classified as hazard type 1 dams (Hanshue & Harrington, 2011). The Moores Park Dam is classified as a type 1 hazard rating, meaning that dam failure would result in the loss of life (Hanshue & Harrington, 2011). The hazard classification for the North Lansing Dam is not disclosed in the Draft Grand River Assessment.

The City of Grand Rapids has proposed to remove dams in the downtown area, including the aging Sixth Street Dam. This structure was built in 1917 for hydroelectric purposes, but no longer functions in this role (Watkins & Bowers, 2014). Removing the dam would serve to restore the rapids setting along the Grand River while improving fish habitat and the aesthetic appeal of the riverfront for recreational tourism opportunities (Bunte, 2015). Significant public support has been brought about for this removal by Grand Rapids Whitewater, a coalition that is one of the major proponents of dam removal and riverfront revitalization in Grand Rapids (Grand Rapids Whitewater, n.d.).

Municipal and state support for the riverfront revitalization project in Grand Rapids has gained greater traction in recent years. In 2013, Michigan Department of Natural Resources (DNR) Director, Keith Creagh, discussed political and fiscal support for the roughly $27.5 million project. The Natural Resources Trust Fund and the Great Lakes Fisheries Trust have both been proposed as funding sources, in addition to federal and state grants (Harger, 2013).

A Draft Grand River Assessment by Hanshue & Harrington (2011) projected a decrease in stream flow magnitude and a decrease in flooding risks. Impoundment removal would likely improve downstream transport of large woody debris (LWD) during higher flow events. Furthermore, the removal would likely lower baseline water levels behind the structure and increase channel carrying capacity. The assessment findings in Grand Rapids offered a potential outcome for dam removal upstream in Lansing.

**General Dam Research Trends**

Watershed modeling of removal is difficult because of the unique, dynamic morphology found within individual watersheds. A study by Rumschlag & Peck (2007) of the Munroe Falls Dam removal on the Cuyahoga River in Ohio acquired environmental data before and after dam removal by utilizing cross-sections near the dam reach to record discharge, bedload composition, and bedload depth. These authors noted that the study should not be used as an analog for all dam removal studies, as each river has unique sediment, bedrock, slope, and discharge components (Rumschlag & Peck, 2007).

Despite the difficulty in producing broad models, more dam removal studies have harnessed advancements in geospatial technologies. The geomorphology of the Huron River in north-central Ohio has evolved in response to construction of the Coho Dam in 1969, removal of its spillway in 1994, and complete removal of the dam in 2002. Evans et al. (2007) utilized USGS stream gage data, eight sets of aerial photographs from 1958-2003 (georeferenced and projected to proper UTM coordinate system), and shapefiles of stream bedforms in ArcGIS. Removal of the spillway resulted in a release of sediment from a zone of accumulation behind the spillway, and a decrease in downstream channel sinuosity because of the cutting of chute channels. The channel incision relates to the behavior of point bars, or accumulations of sediment, following the removal of the spillway. Following removal, the centers of point bars tended to migrate towards the outer banks of the river, indicating increased channel incision along the inner bank (Evans et al., 2007).

**Models for Dam Removal Scenarios**

Several multivariate models have been implemented to model the response of a river to the presence of a dam. The Indicators of Hydrologic Alteration (IHA) model is a geospatial technology that generates indices for hydrologic regimes. These indices are based on magnitude and duration of extreme events, timing of extreme events, frequency, and time of high/low pulses, and rate and frequency of condition changes. Impoundment is found to reduce the discharge of 1-day flows most severely, with a less pronounced effect on 90-day flows, indicating that the impact on flow becomes more consistent with increased flow duration (Magilligan & Nislow, 2005).

A 2005 study evaluated four aging dams on the Kalamazoo River between Plainwell and Allegan, MI, all of which were in disrepair and under consideration by the U.S. EPA and Michigan DEQ for removal to restore the natural river. The Spatially Explicit Delivery Model (SEDMOD), a mathematical sediment transport model that simulates streamflow and sediment transport in a channel, was implemented over 730 days with the dams in place, based on flows during flooding in 1947 with dams and without dams in place. Sediment transport simulations reflected a dynamic equilibrium state, and the absence of dams would lower the channel head, promoting further stream erosion and sediment transport (Syed et al., 2005).

A study of an 8.8 km stretch of the Kalamazoo River between Plainwell and Otsego, MI, where two low-head dams are being evaluated for removal by the state of Michigan, offered several options for the assessment of outcomes related to dam removal (Wells et al., 2007). This reach of the river was evaluated for erosion, transport, and deposition of sediments over a 17.7-year period using the CONCEPTS model, along with additional data from channel surveys, sediment cores, and particle-size analysis for channel materials performed by the USGS. Under a dams-out scenario, bed erosion and sediment transport would greatly increase, headlined by a 187% increase in average annual sediment load (Wells et al., 2007).

Both SEDMOD and CONCEPTS offer modeling options for studying the effects of dams on hydrological processes. However, the Soil and Water Assessment Tool (SWAT) was selected as the preferred model because of its ability to account for land use patterns and for its user-friendly interface in ArcGIS.

***SWAT Overview***

The Soil and Water Assessment Tool (SWAT) provides one of the best methods for modeling changes in hydrological basins. This software was developed by the United States Department of Agriculture (USDA) to analyze and predict impacts of land use practices and changes on watersheds (Gassman et al., 2007). SWAT requires data for land use/land cover, weather, soils, topographic relief, and watershed outlets. Data are obtainable through government GIS archives or can be extracted from data tables within the model (Arnold et al., 1998).

SWAT uses rigorous algorithms and operates on a continuous daily time step to simulate hydrologic balance in a watershed. The model emphasizes incorporating land use change and water quality data to approximate actual future conditions (Arnold et al., 1998). SWAT has been used to predict chemical yields, sedimentation patterns, and streamflow magnitude, among other predictors.

The basic structure for SWAT includes hydrologic, land management, and soil parameters. These inputs are used to divide a watershed into sub-watersheds, or Hydrologic Response Units (HRUs), based on shared characteristics such as land use/management, sub-watershed area, and soil type. HRUs are very important in model calibration and validation where the most impactful parameters on model output are determined (Arnold et al., 1998). SWAT has ongoing limitations associated with algorithm development which include erosion and sediment routing algorithms, subsurface tile drainage algorithms, modeling of nutrient cycling, uncertainty analyses, and modeling intended to reflect real world hydrologic scenarios and data (Gassman et al., 2014).

A study of the Huron and Raisin River watersheds of southeast Michigan used SWAT to analyze the influence of impoundments, including for stream nutrient transport (Bosch, 2008). These watersheds represented differing degrees of dam influence, with 88 dams in the Huron River Watershed and 14 dams in the Raisin River Watershed. The simulation process for stream discharge in both watersheds included calibration with daily measurements on record with the USGS from 1998-2001, and validation from 2002-2005. Simulation data more strongly correlated to observed data for the Raisin River Watershed than the Huron River Watershed, largely due to greater availability of observed data for the Raisin River Watershed. Both models showed an increase in nitrogen and phosphorus loads in the absence of impoundments, with the most noticeable change near river mouths or high runoff source areas. More specifically, the Raisin River watershed model underpredicted discharge against daily and monthly records, while the Huron River watershed model overpredicted monthly discharge and underpredicted daily discharge. Furthermore, simulated stream flow during the validation period was consistently overpredicted during the summer, when flow magnitudes are typically lower (Bosch, 2008).

While SWAT has inherent limitations, it is a useful model for representing the real-world hydrologic conditions in a watershed. The model output tends to be more accurate with greater availability of observed data for streamflow, sediment, soil, land use, and climatology characteristics. The next chapter discusses the methodological approach of SWAT setup and calibration, and issues throughout the research.

CHAPTER III

METHODOLOGY

**Study Area – Grand River Watershed**

The Grand River is the longest river in Michigan, and the Grand River Watershed represents the second largest watershed in Michigan, only behind the Saginaw River Watershed. The location of the watershed, in respect to Michigan and the Great Lakes region, is shown in Figure 3.

Lansing lies along the middle segment of the Grand River Watershed. This segment has a continental climate pattern with an average annual precipitation of 34 inches and average annual snowfall of roughly 40 inches (Hanshue & Harrington, 2011).

A combination of groundwater characteristics and surface flow characteristics dictate stream discharge. The USGS attributes uncertainty in discharge-frequency estimates to basin fluxes such as soil permeability, channel slope, and mean annual precipitation (Perry, 2008). The flow pattern of the Grand River varies seasonally, yet predictably. Flows of greater magnitude correspond to heavier spring and early summer precipitation with saturated soils and snow melt, along with seasonal fall rains and plants ceasing transpiration processes. Flows of lower magnitude correspond to lessening precipitation in late summer and less winter infiltration and runoff with precipitation stored as snow and ice (Hanshue & Harrington, 2011).

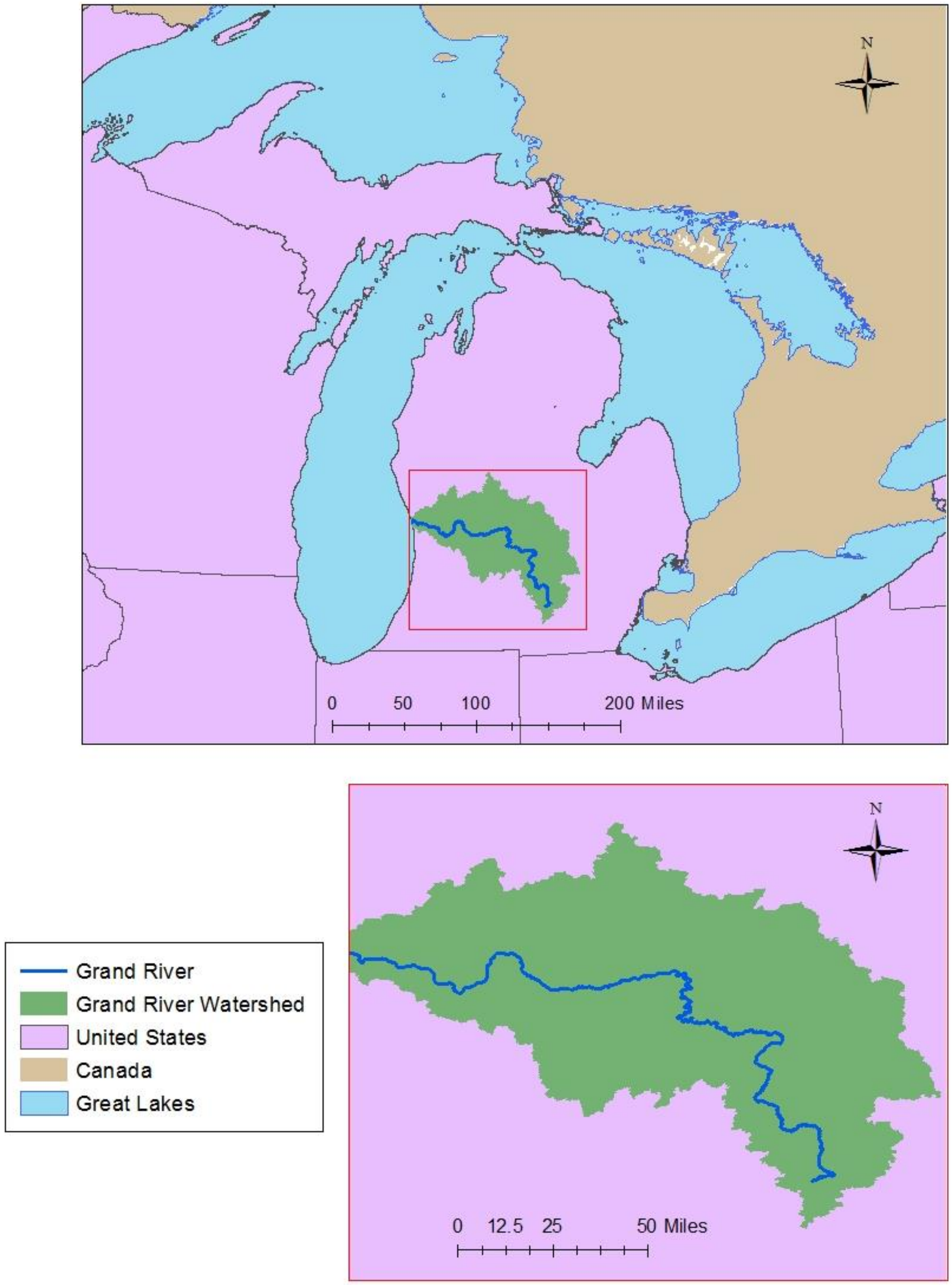


*Figure 3: Grand River Watershed Locator Map*

Table 2 provides the average monthly discharge in the Grand River at the Lansing stream gauge, maintained by the USGS. The stream gauge in Lansing is located downstream of both the Moores Park and North Lansing Dams. Any streamflow that passes through these dams, therefore, must also pass through the stream gauge.

*Table 2: Average Monthly Discharge at Lansing, Michigan*

| Month | Discharge ($ft^3/s$) 2004-2008 | Discharge ($ft^3/s$) 2009-2013 |
|---|---|---|
| **January** | 1753 | 984 |
| **February** | 1369 | 1242 |
| **March** | 1999 | 2057 |
| **April** | 1406 | 2107 |
| **May** | 1409 | 1875 |
| **June** | 1113 | 1369 |
| **July** | 592 | 663 |
| **August** | 384 | 642 |
| **September** | 709 | 377 |
| **October** | 517 | 494 |
| **November** | 704 | 634 |
| **December** | 1255 | 950 |

*Source:* United States Geological Survey (USGS), 2016.

The Moores Park Dam is the most upstream of the two dams in this study (Figure 2), and is listed as a high-risk structure by the City of Lansing. The risk classification of dams in Lansing does not relate to the condition of the structure or likelihood of the structure failing, but rather the impact if the structure was to fall (Dam Failure, n.d.). This impoundment was originally constructed in 1904 with the intention of producing hydroelectric power for an adjacent power plant. Hydroelectric power is no longer produced by this dam, but the water behind the structure is used to cool turbines used to produce electricity from the power plant (Dam Failure, n.d.).

The current North Lansing Dam was constructed in 1936, and is listed as a moderate-risk structure by the City of Lansing. The same risk criteria for classifying the risk of the Moores Park Dam is used to classify the risk of the North Lansing Dam. The original dam at this location, put in place in 1838, was comprised of earthen material; however, the structure was breached in 1844 and subsequently rebuilt, marking the only dam failure to occur at this site. In conditions where flooding is of concern to the surrounding area, the dam can be opened to ease river flows, pending approval from the Michigan Department of Environmental Quality (MDEQ) (Dam Failure, n.d.).

Effective mitigation for dam failure, per Lansing city officials, involves removal of both dams (Dam Failure, n.d.). These dams are significant in size and would likely require redevelopment of the surrounding riverfront if they are removed. The removal of the North Lansing Dam presents the most significant impact to its surrounding area. Since the riverfront development surrounding the North Lansing Dam is based upon the imprint of the pond behind the dam, infrastructure redevelopment to the Riverwalk and storm drain system would likely be required (Dam Failure, n.d.).

**Methodological Approach**

Modeling utilized the Soil and Water Assessment Tool (SWAT) and a Geographic Information System (GIS). Part of the utility of SWAT is its ability to account for land use changed through time (Arnold et al., 1998). Bosch (2008) outlines the methods for calibrating SWAT for a watershed. The GIS interface for

SWAT, called ArcSWAT, facilitates GIS data input into the model. The objective was to import watershed boundaries and outlets, impoundment characteristics, weather data, topography, and soil types into the model. Initial model parameterization necessitated importing data for the entire Grand River watershed; however, the study area of greatest concern was focused on a 5 km reach upstream of the Moores Park Dam to a 5 km reach downstream of the North Lansing Dam. The determination of the spatial extent of this study area referred to that of Rumschlag & Peck (2007) as the greatest magnitude of river morphology was within 5 km upstream and downstream of the Munroe Falls Dam on the Cuyahoga River following impoundment removal.

Much of the input data to construct a watershed model with SWAT was obtained via the Michigan Center for Geographic Information (MiCGI). The MiCGI maintains watershed shapefiles for river basins throughout Michigan, including the Grand River. Information including soil types, a 90-meter spatial resolution Digital Elevation Model (DEM) for the state of Michigan, and land cover data can all be imported into ArcGIS from the MiCGI and clipped to the Grand River watershed boundary. Figure 4 is a schematic diagram for SWAT model simulation.

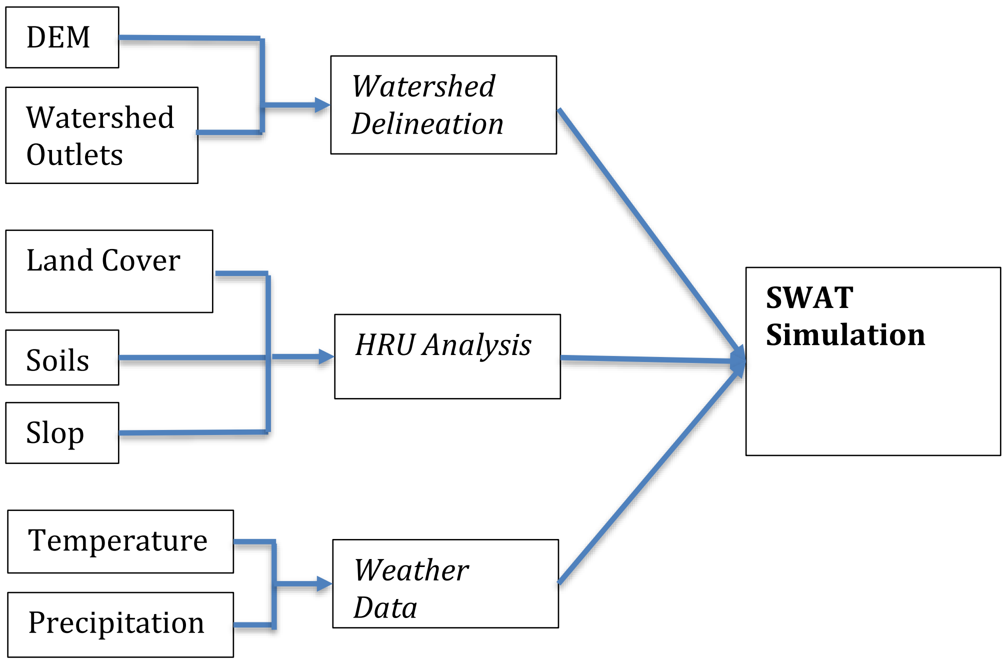


*Figure 4: Diagram for SWAT Model Simulation*

**SWAT Model Setup**

There are three main components to SWAT model construction: Watershed delineation, HRU Analysis, and Weather Data Definition. Watershed delineation involved setting the watershed boundary, importing an elevation profile, and defining watershed outlets. Watershed boundary data were available via the HUC-8 sub-watershed boundaries provided by the USDA. The DEM for Michigan was clipped to the watershed boundary, as shown in Figure 5. Watershed outlets were defined in ArcSWAT through analyzing the DEM.

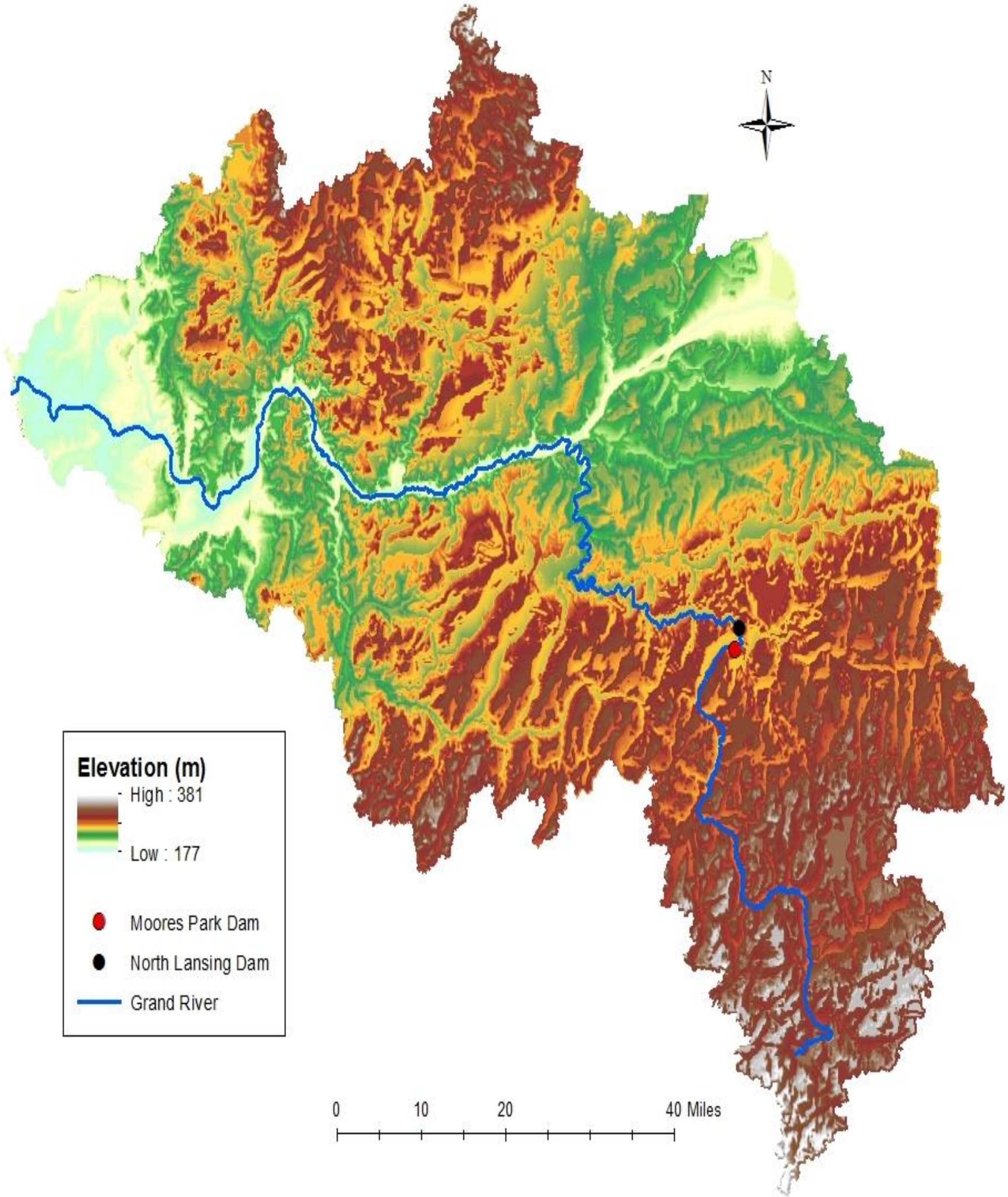


*Figure 5: DEM of Grand River Watershed*

Much of the watershed has little variation in elevation. This is consistent with the relatively flat topography of the lower portion of the Lower Peninsula of Michigan. A slope map representing the percentage of topographic relief throughout the watershed, as compiled from the DEM, is shown in Figure 6.

HRU analysis combined layers for land use/land cover, major soil types, and watershed slopes. HRUs represented modeled soil/land use/management combinations within a sub-watershed, and are represented as a percentage of the watershed area. For ArcSWAT, sub-watershed delineation was utilized to divide the watershed based on topographic features. Once this occurs, either a single soil/land use/management scenario may be modeled, or the sub-watershed may be divided into multiple HRUs. More information regarding watershed configuration is available in Appendix B of the SWAT Input/output Documentation (Arnold et al., 2012a).

Land cover data (30-meter spatial resolution) was obtained from the USGS Land Cover Institute's (LCI) National Land Cover Database (NLCD). The U.S. land cover database shapefile for NLCD 2011 was imported into ArcGIS, clipped to the watershed boundary, converted to raster data, and reclassified to combine irrelevant features into the same class.

Soil data were available for Michigan through the MiCGI. As with the watershed DEM, the imported soils shapefile for the entire state was clipped to the watershed boundary. Slope data for the watershed were derived from the watershed DEM in ArcGIS.

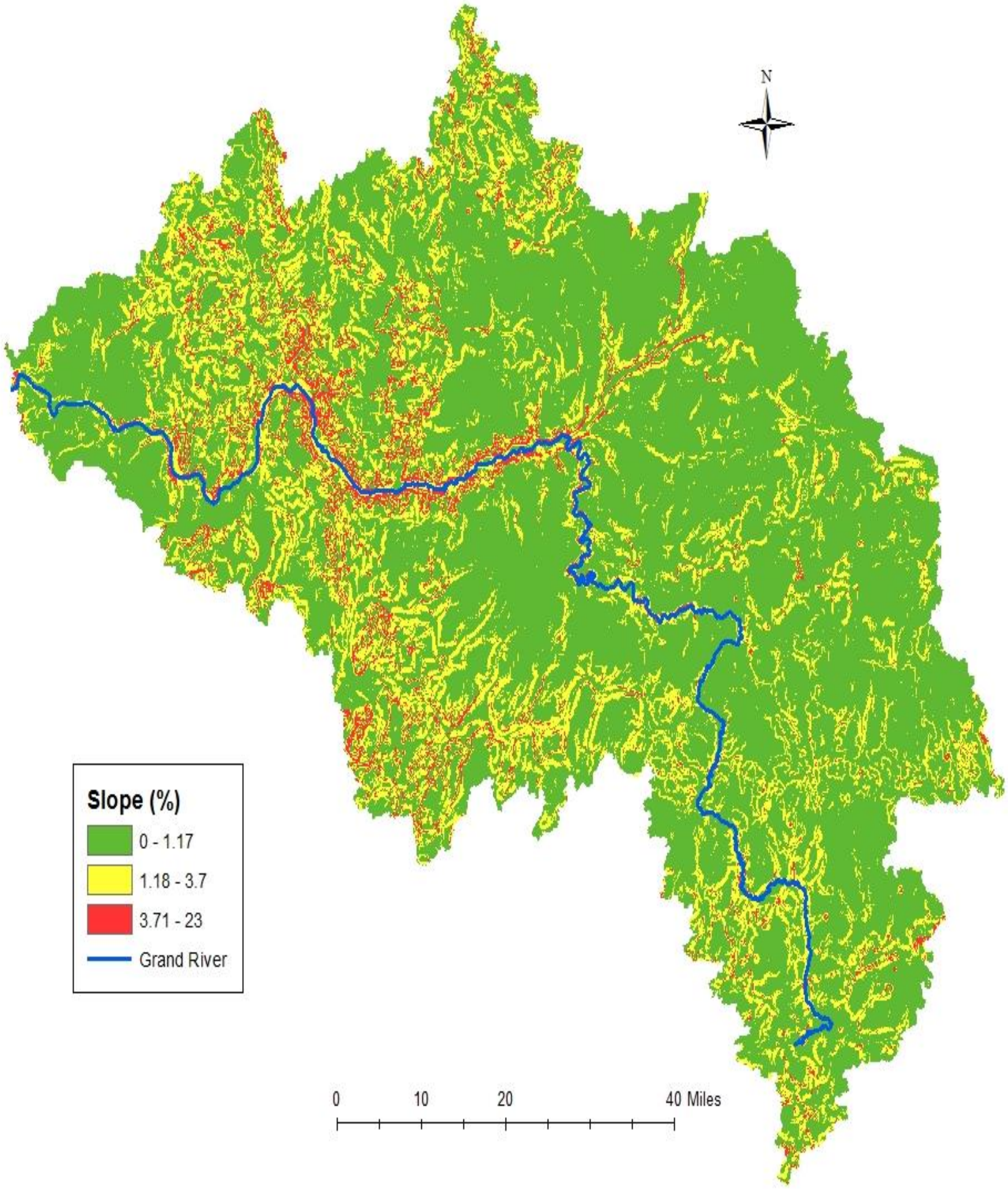


*Figure 6: Slope Map of Grand River Watershed*

Forming a SWAT simulation necessitated the inclusion of weather data, most importantly, temperature and precipitation data. Climate data were extracted from weather stations in the watershed from the Global Weather Data for SWAT website (Global Weather Data for SWAT, 2017). Attainable variables included temperature (°C), precipitation (mm), wind (m/s), relative humidity (percent), and solar radiation ($MJ/m^2$).

The input of impoundment characteristics is discussed within the SWAT Input/Output documentation (Arnold et al., 2012a). Chapter 1 of the documentation mentions that water bodies on the stream network of the watershed are represented within SWAT as reservoirs or ponds. Impoundment characteristics may be input as a reservoir or pond depending on the location of the dam with respect to the main channel or other channels, and the size of the impoundment. I chose to simulate the study dams in the watershed as reservoirs.

Table 3 provides information from the U.S. Army Corps of Engineers (USACE) National Dam Inventory pertaining to the size, storage capacity, and other notable attributes of the Moores Park and North Lansing Dams. The information was used in SWAT to simulate the placement of the impoundments within the watershed. Similar information was obtained from the National Dam Inventory for other dams in the delineated watershed.

*Table 3: Moores Park and North Lansing Dams Data*

| | *Moores Park Dam* | *North Lansing Dam* |
|---|---|---|
| **Completed** | 1904 | 1936 |
| **Latitude** | 42.7184 | 42.75 |
| **Longitude** | -84.5608 | -84.55 |
| **Maximum Storage** | 2140 acre-ft. | 1810 acre-ft. |
| **Normal Storage** | 1928 acre-ft. | 500 acre-ft. |
| **Maximum Discharge** | 10300 $ft^3/s$ | 17500 $ft^3/s$ |
| **Dam Height** | 22.1 ft. | 20 ft. |
| **Hydraulic Height** | 23.89 ft. | 12 ft. |
| **Length** | 473.39 ft. | 252 ft. |
| **Drain Area** | 768 $mi^2$ | 1230 $mi^2$ |
| **Surface Area** | 310 acres | 92 acres |

*Source:* USACE, United States Army Corps of Engineers, 2016.

**Data Manipulation**

The model setup initially involved delineating the watershed using the Automatic Watershed Delineation Tool in SWAT. The DEM for the watershed was imported and analyzed by the model to estimate the flow direction and flow accumulation of the watershed stream network. Following this, the model required information for the minimum area of each HRU in the watershed to create the stream network and outlets. I selected 3572 hectares per HRU as the minimum size to depict the frequency and extent of streams in the watershed.

Once the stream network was created, watershed outlets were defined. I manually added watershed outlets for the Moores Park and North Lansing Dams, along with all other dams in the upper portion of the watershed. This was important because the SWAT program will only allow for placement of the dams and reservoirs at HRU outlets or user-defined outlets.

The USGS stream gauge at Lansing was selected as the whole watershed outlet, as all flows that contribute to the dam study area also contribute to this location. Additionally, the reduction in watershed size minimized the size of the files to be processed in model simulation and allowed SWAT to better process the model output.

The final steps in the Automatic Watershed Delineation were to delineate the watershed, calculate subbasin parameters, and manually add reservoirs to represent dam locations. A total of eight reservoirs were added to the basin at the user-defined subbasin outlets, including those for the Moores Park and North Lansing Dams. The delineated watershed had a minimum elevation of 249m and a maximum elevation of 350m (*sd* = 12.43).

Land use, soils, and slope definitions were reclassified and defined using the HRU Analysis window. This process divided the watershed into more unique sub-watersheds that contribute to the overall flow of water in the system.

The NLCD 2011 land cover file was imported for land use definition. The SWAT program reclassified the file for the delineated watershed. Table 4 reflects the types and distribution of land use in the watershed following reclassification. Most of the land in the watershed pertains to agricultural land – row crops, hay, and forest – deciduous.

*Table 4: Land Use Classification*

| Land Use | Abbrev. | Area (ha) | Area (acres) | % Watershed Area |
|---|---|---|---|---|
| Agricultural Land – Row Crops | AGRR | 100,917 | 249,370 | 33.97 |
| Forest – Deciduous | FRSD | 44,268 | 109,388 | 14.90 |
| Hay | HAY | 71,134 | 175,775 | 23.95 |
| Industrial | UIDU | 1,334 | 3,297 | 0.45 |
| Residential – Low Density | URLD | 13,138 | 32,465 | 4.42 |
| Residential – Medium Density | URMD | 11,457 | 28,311 | 3.86 |
| Residential – High Density | URHD | 4,384 | 10,834 | 1.48 |
| Water | WATR | 68 | 168 | 0.02 |
| Wetlands – Forested | WETF | 49,574 | 122,500 | 16.69 |
| Wetlands – Non-forested | WETN | 770 | 1,904 | 0.26 |

Soil classifications were based on the STATSGO soils file obtained from the MiCGI. The HRU Analysis yielded 11 different soil orders with varying area, as shown in Table 5. Loam represents the dominant surficial material found in the watershed.

*Table 5: Soil Orders by Area*

| Soil Code | Area (ha) | Area (acres) | % Watershed Area |
|---|---|---|---|
| MI010 | 10,848 | 26,807 | 3.65 |
| MI014 | 44,613 | 110,242 | 15.02 |
| MI017 | 16,835 | 41,601 | 5.67 |
| MI018 | 3,377 | 8,345 | 1.14 |
| MI022 | 22,945 | 56,699 | 7.72 |
| MI024 | 4,635 | 11,454 | 1.56 |
| MI029 | 4,501 | 11,122 | 1.52 |
| MI034 | 81,496 | 201,381 | 27.44 |
| MI035 | 29,441 | 72,750 | 9.91 |
| MI036 | 37,726 | 93,224 | 12.70 |
| MI061 | 40,624 | 100,385 | 13.68 |

Slopes for the watershed were divided into three classes: 0-3%, 3-6%, and greater than 6% slope. The DEM (Figure 5) and the slope map (Figure 6) both demonstrate the degree to which the watershed does not have high topographic relief. The slope classes are described in Table 5.

*Table 6: Slope Classification*

| % Slope | Area (ha) | Area (acres) | % Watershed Area |
|---|---|---|---|
| 0-3 | 294,483 | 727,683 | 99.14 |
| 3-6 | 2,478 | 6,124 | 0.83 |
| 6-9999 | 82 | 203 | 0.03 |

After completing the HRU analysis, I set the threshold percentages for each layer. The minimum levels for each hydrologic response unit were set to 10% for land use, soil class, and slope class. These thresholds, which followed those used in modeling the Kalamazoo River Watershed (Serfas, 2012), resulted in the distribution of 521 HRUs and 63 subbasins throughout the watershed.

The last step before running SWAT and beginning calibration was to write the database input tables for weather, soil, water use, groundwater, channel, management, and configuration files. Once database files were generated, the initial SWAT model ran from January 1, 2000, until December 31, 2013. The first four years were used as the recommended warm-up period for the model.

**SWAT Calibration and Validation**

A local sensitivity analysis preceded SWAT calibration and validation. This process identified the rate of change in model output because of model inputs, or parameters (Arnold et al., 2012b). This analysis served to make the model more reliable when making predictions without data to validate. Calibration and validation success was determined based on statistical goodness of fit. The statistical methods utilized in this research are discussed later in this chapter.

Local sensitivity analysis involved the manipulation of values individually (Arnold et al., 2012b). This was done using the Manual Calibration Helper window in SWAT, which allows for multiplying a parameter by a threshold, adding to a parameter by a threshold, or replacement of the parameter value. The SWAT Input/Output documentation file booklet describes the variables that may contribute most greatly to simulated watershed characteristics (Arnold et al., 2012a).

The SWAT Calibration Techniques Manual suggests that the user calibrate the hydrology of the delineated watershed before calibrating the sediment and water quality parameters (Arnold et al., 2012b). Since this research focused on modeling stream discharge, I calibrated and validated the model for hydrology only.

Monthly average stream discharge data were applied during calibration and validation. These data were available from the USGS. The USGS maintains 21 stream gauges in this watershed, offering varying data availability, data coverage, and temporal span of records. The primary location of concern for stream discharge was Lansing. This stream gauge was utilized since this location is downstream of both dams and is the whole watershed outlet of the delineated watershed.

The calibration and validation of the model compared simulated discharge values to observed discharge values for the Lansing USGS gauging station. Monthly mean discharge from the USGS is expressed in cubic feet per second ($ft^3/s$), while total water yield in SWAT is expressed as a depth per month in millimeters (mm). To account for the difference in units, USGS discharge data were converted to the SWAT format using Equation 1 and Equation 2. The model determined that the area of the delineated watershed was approximately 2970 $km^2$, or $2.97*10^9$ m.

Equation 1: Conversion from $ft^3/s$ to $m^3/s$

$$1\ ft^3/s = 0.0283168\ m^3/s$$

Equation 2: Conversion from $m^3/s$ to mm

$$Q\ (mm) = \frac{\left(Q\left(\frac{m^3}{s}\right)\right) * 1000 * 24 * (\#days\ in\ month) * 3600}{watershed\ area\ (m^2)}$$

Calibration of the "dam-in" scenario used data from January 1, 2004, to December 31, 2008, and validation of the "dam-in" scenario used data from January 1, 2009, to December 31, 2013. This duration was selected with consideration of the complete weather and stream flow records available, and to generate a model more correlated with modern land use/land cover within the watershed.

## Simulation Evaluation Metrics

### *Standard Deviation*

The standard deviation of a dataset refers to the distribution of data from the mean. No guidelines are provided within the SWAT documentation as to an optimal value for the standard deviation. However, a higher standard deviation for simulated data than for observed data would indicate a greater distribution of data from the mean and more outlier values. Generally, a smaller value for the standard deviation is preferred, given that this describes the proximity of data to the mean of the data. While not affected by extreme values, the standard deviation does not provide the full range of data and it assumes that these data are normally distributed.

### *Pearson's Correlation/Coefficient of Determination*

Pearson's correlation (R) can be calculated to determine the relationship between observed data and simulated data. This correlation coefficient ranges from -1 to +1, with values greater than 0.7 signifying a strong positive linear relationship and values less than -0.7 signifying a strong negative linear relationship. The coefficient of determination ($R^2$) accounts for the variance of the dependent variable that can be attributed to the variation in the independent variable (Cronk, 2016). The $R^2$ values range from 0 to 1 and can describe variability in a regression model. An $R^2$ value of 1 indicates perfect collinearity between simulated and observed data.

The Pearson correlation coefficient is determined using z-scores, meaning that both variables ought to be normally distributed. Furthermore, the relationship between the two variables is assumed to be linear. If this assumption is not met, the Spearman *rho* correlation coefficient will be used in place of the Pearson correlation coefficient (Cronk, 2016).

Since the coefficient of determination only quantifies the combined dispersion versus the dispersion of observed and predicted data, the $R^2$ values can be close to 1 even in a model that consistently over-or under-predicts values (Krause et al., 2005). I applied Pearson's correlation coefficient and coefficient of determination to the relationship between modeled and observed stream flow, and the relationship between Dams-In and Dams-Out modeled stream flow.

***Nash-Sutcliffe Efficiency***

The Nash-Sutcliffe Efficiency (NSE) closely resembles Pearson's correlation, and describes the relationship between observed data and simulated data. NSE is concerned with the variance of data, and addresses the ratio of the variance of observed and simulated data to the variance of observed data and its mean (Nash and Sutcliffe, 1970). The NSE is calculated using Equation 3.

Equation 3: Nash-Sutcliffe Efficiency (NSE)

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim})^2}{\sum_{i=1}^{n}(Y_i^{obs} - Y^{mean})^2}\right]$$

In the NSE formula, $Y^{obs}$ is the observed streamflow, $Y^{sim}$ is the SWAT simulated streamflow, and $Y^{mean}$ is the mean of the observed streamflow. NSE ranges from $-\infty$ to 1, with 1 representative of absolute collinearity between the observed data and the simulated data. In general, the acceptable range for NSE values is 0 to 1, with a range of .5 to 1 preferred for SWAT simulations (Moriasi et al., 2007). These guidelines were applied to determining if the model outputs were statistically significant.

The NSE calculates the difference between observed and predicted values as squared values, meaning that larger quantities are overestimated. Thus, the model is overestimated during high-flow events and underestimated during low-flow events. Much like with the $R^2$, NSE is not effective with accounting for errors in model prediction during low-flow events (Krause et al., 2005).

***Root Mean Square Error***

The Root Mean Square Error (RMSE) was used to indicate the amount of error found with simulated data. This error statistic was compared with the Mean Absolute Error (MAE). A lower value of RMSE is preferred, with values closer to zero indicating less error between simulated data and observed data (Santhi et al., 2001).

RMSE is a similar expression to that of standard deviation, is a reliable measure of uncertainty in prediction, and provides a quadratic loss function. However, the presence of extreme values effects the accuracy of this measure (Makridakis & Hibon, 1995). RMSE also is a function of three characteristics and varies with the Mean Absolute Error (MAE), the square root of the number of errors, and the distribution of error magnitudes (Willmott & Matsuura, 2005). The RMSE statistic is determined using Equation 4.

Equation 4: Root Mean Square Error (RMSE)

$$RMSE = \frac{\sqrt{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim})^2}}{n}$$

***Mean Absolute Error***

Another useful error statistic utilized in this study was the Mean Absolute Error (MAE). Summing the magnitudes of errors and dividing by *n*, or the number of total observations, yielded the MAE. Much like with the RMSE and the MSE, the MAE will increase with increasing variance of the frequency distribution of error magnitudes (Willmott & Matsuura, 2005). Unlike the RMSE, the MAE is unambiguous and therefore a more natural measure of average error (Willmott & Matsuura, 2005). The equation for determining the MAE is shown in Equation 6.

Equation 5: Mean Absolute Error (MAE)

$$MAE = \frac{1}{n}\sum_{i=1}^{n} e_i$$

***Percent Bias***

Percent Bias (PBIAS) showed if modeled data were generally overestimated (negative value) or underestimated (positive value) as compared to observed data. As with the prior error metrics, PBIAS is most ideal when the value is closer to 0 – that is, an insignificant difference exists between modeled and observed data (Moriasi et al., 2007). The statistic is determined using Equation 6.

Equation 6: Percent Bias (PBIAS)

$$PBIAS = \left[\frac{\sum_{i=1}^{n}\left(Y_i^{obs} - Y_i^{sim}\right) * (100)}{\sum_{i=1}^{n}(Y_i^{obs})}\right]$$

CHAPTER IV

RESULTS

## Sensitivity Analysis

Initial model runs significantly overestimated water depth (discharge) at the watershed outlet, and overestimated the ratio of surface flow to baseflow into the stream channel. Streams in south-central Michigan typically have a baseflow index of 50-70% (Santhi et al., 2008). Through trial and error with the Manual Calibration window in SWAT, I determined the most sensitive parameters related to watershed hydrology. The most sensitive parameters were: ALPHA_BF, Cn2, ESCO, GWQMN, GW_REVAP, Rchrg_dp, and SOL_AWC.

ALPHA_BF is the baseflow recession constant, or baseflow alpha factor. This constant reflects response to recharge in groundwater flow. A low baseflow recession constant reflects slow response to recharge, while a high baseflow recession constant reflects more rapid response to recharge (Arnold et al., 2012a).

Cn2, or the curve number, represents the surface runoff in a HRU. A higher curve number means that the land use, soils, and land cover combinations yield a high amount of overland runoff into the stream network. Conversely, a low curve number indicates lesser surface runoff and higher baseflow (Serfas. 2012). The average Cn2 for the delineated watershed was approximately 63.

ESCO represents the soil evaporation compensation factor, or the ratio of water in the soil that is lost because of evaporative processes. The ESCO factor ranges from 0 to 1, with 0 being no loss of soil moisture and 1 being complete loss of soil moisture to evaporation. Adjustments to this parameter have a small effect on the amount of surface runoff in the HRU or watershed (Serfas, 2012).

GWQMN is the threshold water depth in the shallow aquifer required for the base flow to occur. This parameter is also referred to as the deep percolation loss. An increase in the GWQMN will offset calibration issues pertaining to high baseflow and low evapotranspiration.

GW_REVAP is the groundwater revap coefficient. Revap refers to the water transfer from the shallow aquifer to the root zone of the soil. Decreasing the parameter increases baseflow, while increasing the parameter increases water transfer to plants and decreases baseflow (Abraham et al., 2007).

Rchrg_dp refers to the deep aquifer percolation fraction. This value dictates the groundwater aquifer height and ranges from 0 to 1, with values closer to 1 increasing the deep aquifer recharge and decreasing the height of the water table. As the deep aquifer recharge value increases, the movement of water into the stream channel decreases (Abraham et al., 2007).

SOL_AWC is the soil available water capacity. If the soil available water capacity is higher, the amount of surface flow relative to baseflow would decrease (Abraham et al., 2007).

Adjustments were made to these hydrological parameters within the range of values that SWAT accepts. These manual calibrations accounted for the water balance of the watershed and did not yield any hydrological warnings in the SWAT Error Checker window. A description of the manually calibrated parameters is outlined in Table 7.

*Table 7: Parameter Adjustments During Manual Calibration*

| Parameter | SWAT Accepted Range | Substituted Value | Land Use |
|---|---|---|---|
| ALPHA_BF | 0-1 | 0.1 | All |
| Cn2 | 10-90 | 55 | AGRR, FRSD |
| Cn2 | 10-90 | 60 | HAY |
| Cn2 | 10-90 | 65 | UIDU, URHD, URLD, URMD, WETN |
| Cn2 | 10-90 | 62 | WETF |
| ESCO | 0-1 | 0.1 | All |
| GWQMN | 0-5000 | 200 | All |
| GW_REVAP | 0.02-0.20 | 0.20 | All |
| Rchrg_dp | 0-1 | 0.5 | All |
| SOL_AWC | 0.1-0.2 | 0.15 | All |

**Scenario One – Dams-In**

The methodology was implemented for two scenarios – one with the Moores Park and North Lansing Dams in place, and one with the dams not in place. SWAT simulations were performed with the same hydrological parameter adjustments, period, and geospatial data in each case. As mentioned in Chapter III, the model was only calibrated and validated for the Dams-In simulation.

A comparison of the observed and calibrated water depth values are shown in Table 8 and in Figure 7. The model accounted for 419.862 mm of the 337.3064 mm of the annual discharge at the USGS gauge in Lansing, yielding an overprediction of roughly 25% (Table 8).

*Table 8: Dams-In Calibration Streamflow Results*

| Month | USGS (mm) 2004-2008 | Calibration (mm) 2004-2008 |
|---|---|---|
| January | 44.7666 | 50.358 |
| February | 34.9594 | 48.182 |
| March | 51.03 | 69.728 |
| April | 35.8988 | 55.292 |
| May | 35.9858 | 46.698 |
| June | 28.4256 | 35.714 |
| July | 15.127 | 26.622 |
| August | 9.8078 | 19.298 |
| September | 18.097 | 16.718 |
| October | 13.1918 | 9.708 |
| November | 17.9818 | 9.1 |
| December | 32.0348 | 32.444 |
| *Σ yearly* | *337.3064* | *419.862* |
| | | *124.5%* |

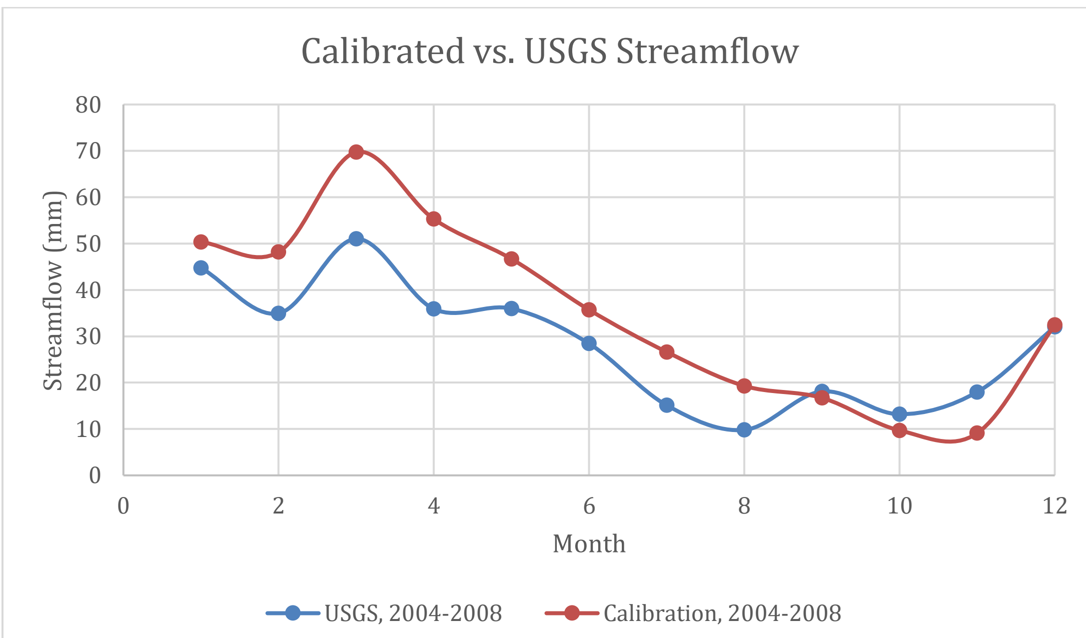


*Figure 7: Dams-In Calibrated Streamflow Results vs. USGS*

A comparison of the observed and validated water depth values in shown in Table 9 and in Figure 8. The model accounted for 426.024 mm of the 342.0372 mm of the annual discharge at the USGS gauge in Lansing, yielding an overprediction of roughly 25% (Table 9).

*Table 9: Dams-In Validated Streamflow Results*

| Month | USGS (mm) 2009-2013 | Validation (mm) 2009-2013 |
|---|---|---|
| January | 25.1264 | 31.162 |
| February | 31.7188 | 42.626 |
| March | 52.516 | 68.546 |
| April | 53.7926 | 59.652 |
| May | 47.879 | 63.448 |
| June | 34.9664 | 49.276 |
| July | 16.939 | 33.874 |
| August | 16.3882 | 22.712 |
| September | 9.627 | 14.652 |
| October | 12.6194 | 12.874 |
| November | 16.1986 | 10.25 |
| December | 24.2658 | 16.952 |
| *Σ yearly* | *342.0372* | *426.024* |
| | | *124.6%* |

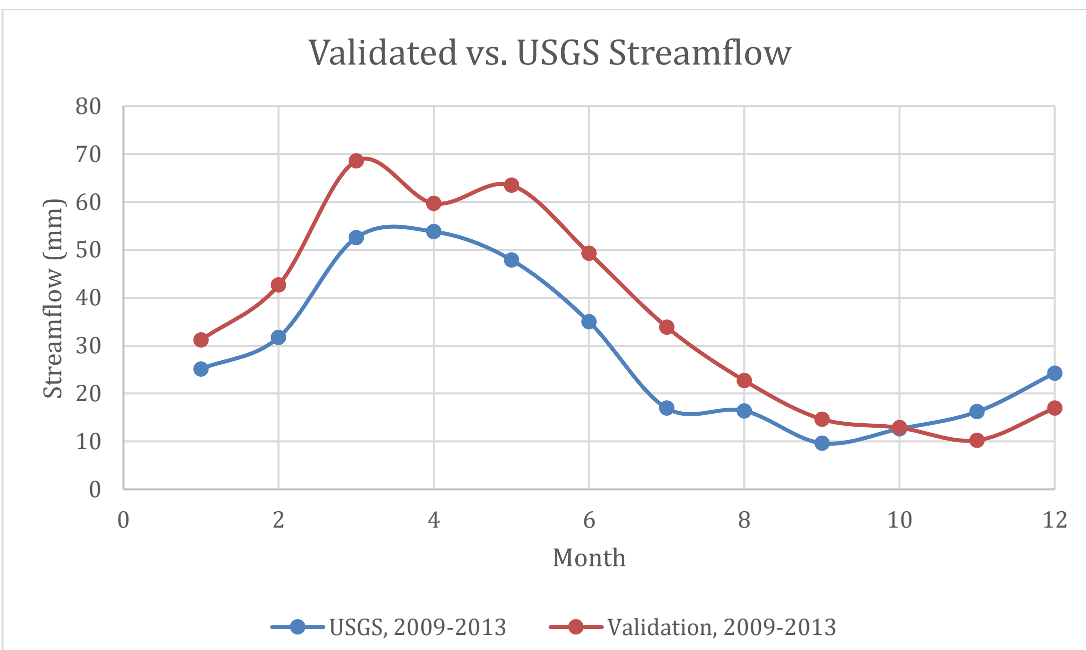

*Figure 8: Dams-In Validated Streamflow Results vs. USGS*

While the simulated streamflow results exceed observed USGS streamflow results during both calibration and validation, the simulated and observed data follow a similar trend throughout the year. The model appears to overestimate higher flow events during peak flooding from January through May, and tends to underestimate lower flow events in late fall and early winter (October-December). The model outputs for calibration and validation also have a greater distribution than the observed data, as indicated by Figures 9 and 10.

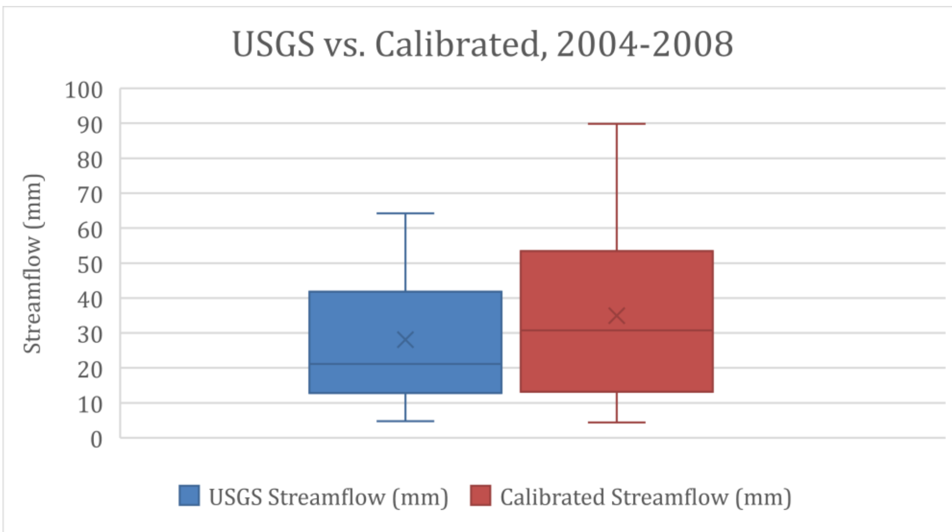


*Figure 9: Box-and-Whisker Plot for Calibration*

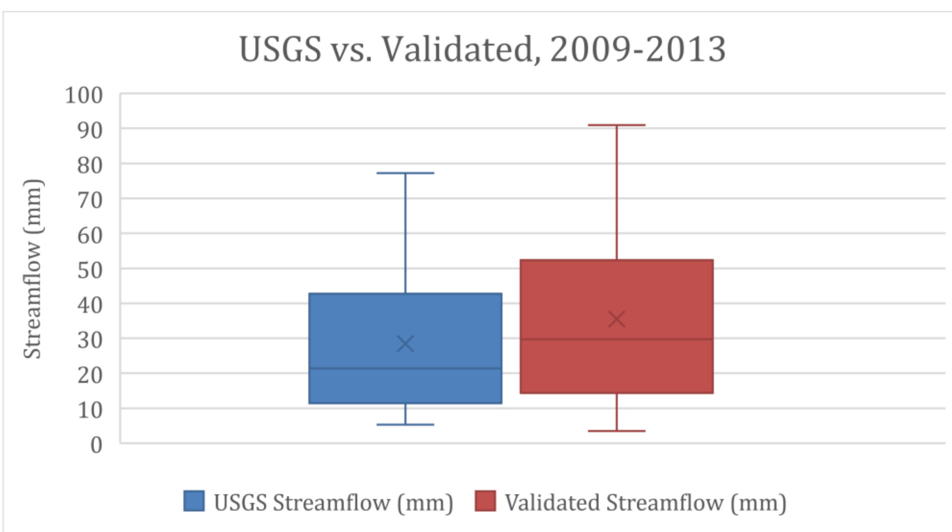


*Figure 10: Box-and-Whisker Plot, Validation*

Further statistical analysis of the model's ability to replicate observed hydrological conditions in the delineated watershed is presented in Table 10. The standard deviation for calibration and validation are rather similar, indicating a similar dispersion of streamflow for individual months.

*Table 10: Dams-In Calibration and Validation Statistics*

| Scenario | Std. Dev. | $R^2$ | NSE | RMSE | MAE | PBIAS |
|---|---|---|---|---|---|---|
| Calibration – Dams-In | 23.848 | 0.854 | -30.755 | 16.186 | 12.743 | -35.233 |
| Validation – Dams-In | 24.950 | 0.880 | -34.955 | 16.309 | 11.721 | -34.933 |

A Pearson's coefficient of determination was found for the correlation between calibrated streamflow and observed streamflow (Table 10). The analysis indicated a strong positive correlation ($R^2$ = 0.854). Simulated higher flow months tended to correlate with observed higher flow months, while simulated lower flow months correspond with observed lower flow months.

A Pearson's coefficient of determination was found for the correlation between validated streamflow and observed streamflow (Table 10). The analysis indicated a strong positive correlation ($R^2$ = 0.880). As with the calibration scenario, simulated higher flow months tended to correlate with observed higher flow months, while simulated lower flow months correspond with observed lower flow months.

The Nash-Sutcliffe Efficiency compared the relationships between calibrated data and observed data, and between validated data and observed data (Table 10). In both scenarios, the NSE statistic indicated that the mean of the observed data is a better predictor of streamflow than the model outputs.

The Root Mean Square Error was applied to determine the amount of error associated with calibrated and validated streamflow predictions (Table 10). RMSE statistics were similar for the calibration and validation scenarios, indicating considerable uncertainty in predicting the streamflow over the course of the year. However, the presence of outliers in the simulated data may have skewed each statistic. While RMSE is an effective measure of model uncertainty, it is heavily skewed by large error magnitudes in predicted data.

The magnitude of errors associated with calibrated and validated data was determined using Mean Absolute Error (MAE, Table 10). As with the Root Mean Square Error, the MAE values for the calibration and validation scenarios were similar, indicating noticeable variance of the frequency distribution of error magnitudes. This value for the distribution of error magnitudes is arguably more reliable than the RMSE value, since outlier data does not significantly impact this statistic.

A Percent Bias statistic determined the average tendency of calibrated and validated data as compared with observed data (Table 10). In each scenario, a negative PBIAS value was derived, meaning that calibrated and validated data were overpredicted by roughly 35%.

While many individual months could be calibrated and validated with statistical significance, the overall trend throughout the year was that the model overpredicted streamflow too much. I generated calibration and validation statistics in Table 11 and Table 12 to separate data into the four seasons. The grouping of months were: *Winter*: January-March; *Spring*: April-June; *Summer*: July-September; and *Fall*: October-December. These groupings reflected the season that more than half of each month belonged to.

*Table 11: Calibration Statistics by Season*

| | $R^2$ | NSE | PBIAS |
|---|---|---|---|
| Winter | 0.719 | 0.005 | -33.251 |
| Spring | 0.8 | -119.937 | -52.267 |
| Summer | 0.008 | -0.234 | -72.247 |
| Fall | 0.923 | -2.853 | 16.833 |

*Table 12: Validation Statistics by Season.*

| | $R^2$ | NSE | PBIAS |
|---|---|---|---|
| Winter | 0.995 | -112.359 | -35.466 |
| Spring | 0.705 | -20.747 | -39.384 |
| Summer | 0.723 | -6.247 | -81.466 |
| Fall | 0.581 | -0.466 | 16.585 |

A Pearson's coefficient of determination was found for the correlation between calibrated streamflow and observed streamflow for each season (Table 11). The analysis indicated a strong positive correlation for each season, except for the summer ($R^2 > 0.7$). During the summer, the model significantly overpredicted streamflow in July but only marginally overpredicted streamflow in August, despite both months having a similar average streamflow (Table 8). During the fall, the model nearly predicted the average streamflow for December, and underpredicted the average streamflow for October and November (Table 8).

A Pearson's coefficient of determination was found for the correlation between validated streamflow and observed streamflow for each season (Table 12). The analysis indicated a strong positive correlation for each season, except for the fall ($R^2 > 0.7$). During the fall, the model nearly predicted the average streamflow for October, and underpredicted the average streamflow for November and December (Table 9). During the winter, the model overpredicted average streamflow by roughly 130% (Table 9).

The Nash-Sutcliffe Efficiency compared the relationships between calibrated data and observed data for each season (Table 11). NSE values for winter and summer were close to 0, indicating that model predictions for average streamflow were roughly as accurate as predictions based on the observed monthly average. The spring NSE value (Table 11) heavily skewed the yearly NSE value (Table 10), as several individual summer months had large overestimations of average monthly streamflow. NSE calculated the difference between observed and predicted values as squared values, leading to overestimations of large quantities.

The Nash-Sutcliffe Efficiency compared the relationships between validated data and observed data for each season (Table 12). Only the fall had an NSE value close to 0, indicating that model predictions for average streamflow were roughly as accurate as predictions based on the observed monthly average. The winter NSE value (Table 12) heavily skewed the yearly NSE value (Table 10), as several individual winter months had large overestimations of average monthly streamflow.

Percent Bias determined the average tendency of calibrated data as compared with observed data for each season (Table 11). In every season besides the fall, a negative PBIAS value was derived, meaning that calibrated data were overpredicted. The positive PBIAS value in the fall indicates that the model underpredicted average monthly streamflow.

Percent Bias determined the average tendency of validated data as compared with observed data for each season (Table 12). A negative PBIAS value was derived for each season besides the fall, meaning that calibrated data were overpredicted. The positive PBIAS value in the fall indicates that the model underpredicted average monthly streamflow. In both the calibration and validation scenarios, the magnitude of the PBIAS value was highest during the summer, indicating more severe overprediction of average monthly streamflow.

**Scenario Two – Dams-Out**

In the Dams-Out scenario, the same input data, hydrological parameters, and simulation timeframe were used in SWAT. The only changes from the Dams-In scenario were not including data for the Moores Park and North Lansing Dams, and not calibrating or validating the model results.

The Dams-In and Dams-Out water depth values are shown in Table 13 and Figure 12. The dams-out model accounted for an annual discharge of 749.222 mm, or a 77.1% increase in annual discharge over the dams-in value of 422.943 mm (Table 13).

*Table 13: Dams-Out Streamflow Results*

| Month | Dams-In, 2004-2013 | Dams-Out, 2004-2013 |
|---|---|---|
| January | 40.76 | 71.433 |
| February | 45.404 | 69.759 |
| March | 69.137 | 87.023 |
| April | 57.472 | 74.41 |
| May | 55.073 | 79.666 |
| June | 42.495 | 71.879 |
| July | 30.248 | 58.152 |
| August | 21.005 | 48.007 |
| September | 15.685 | 45.706 |
| October | 11.291 | 38.524 |
| November | 9.675 | 42.548 |
| December | 24.698 | 62.117 |
| *Σ yearly* | *422.943* | *749.224* |
| | | *177.1%* |

The Dams-Out scenario appeared to overestimate higher flow events more than the Dams-In scenario during peak flooding season from January through May. The Dams-Out and Dams-In scenarios both have a greater distribution of data than the observed data, as indicated by Figure 11.

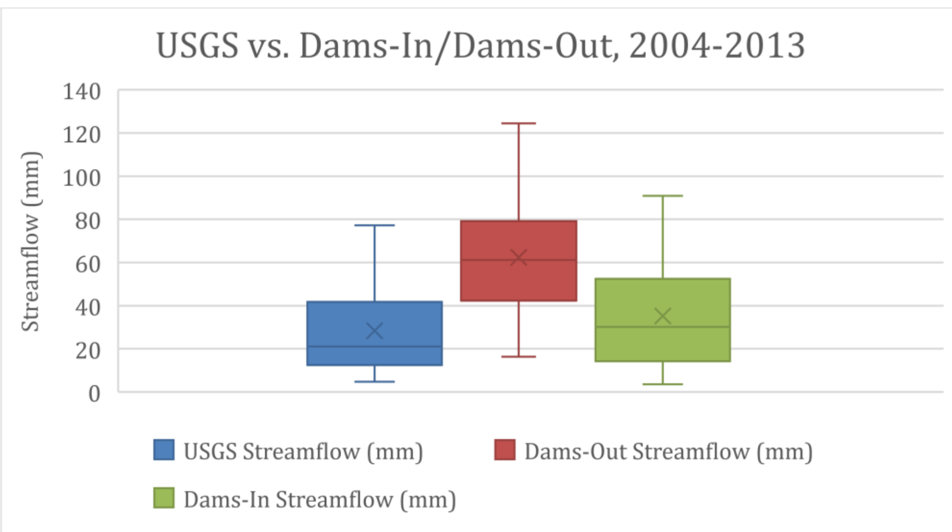


*Figure 11: Box-and-Whisker Plot, Dams-Out*

Further statistical analysis of the Dams-Out model's change from the Dams-In model is presented in Table 14. Additionally, statistical analysis of the Dams-Out model's change from the USGS observed data is presented in Table 15.

*Table 14: Dams-In vs. Dams-Out Statistics*

| Scenario | Mean | Std. Dev. | $R^2$ |
|---|---|---|---|
| Dams-In | 35.245 | 24.304 | 0.936 |
| Dams-Out | 62.435 | 24.041 | |

*Table 15: USGS vs. Dams-Out Statistics*

| Scenario | Mean | Std. Dev. | $R^2$ |
|---|---|---|---|
| USGS | 28.306 | 18.870 | 0.898 |
| Dams-Out | 62.435 | 24.041 | |

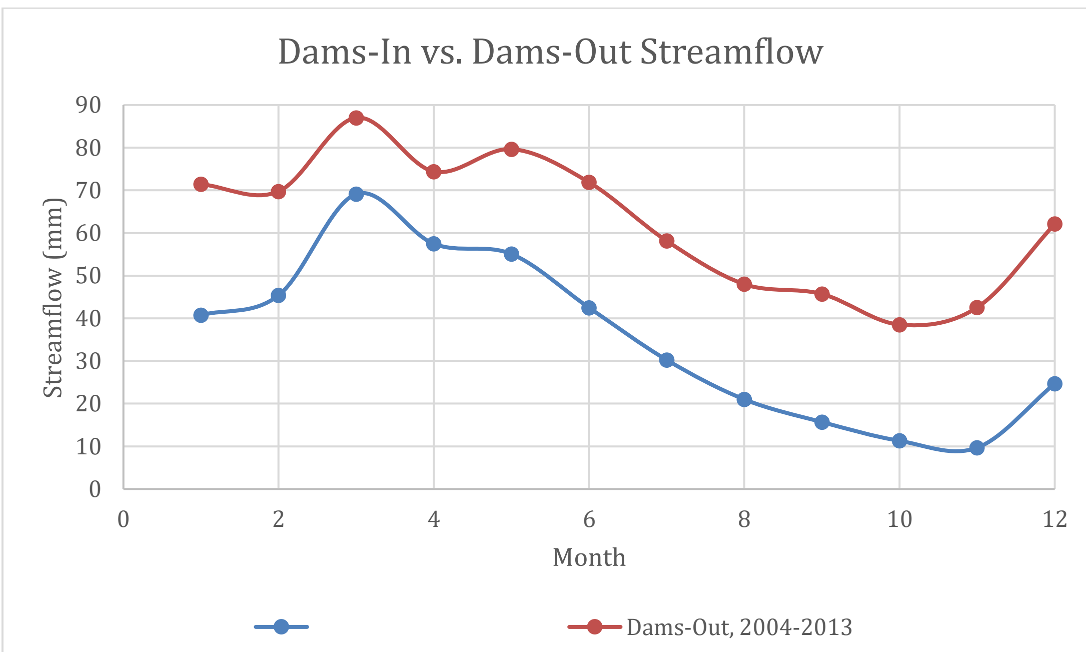


*Figure 12: Dams-In vs. Dams-Out Streamflow*

The standard deviation for Dams-In and Dams-Out was rather similar, indicating a similar dispersion of streamflow for individual months. However, the average monthly streamflow for Dams-Out was nearly double the average monthly streamflow for Dams-In, indicating a significant increase in average streamflow at the Lansing gauge with the Moores Park and North Lansing Dams removed.

A Pearson's coefficient of determination was found for the correlation between Dams-In streamflow and Dams-Out streamflow (Table 14). The analysis indicated a strong positive correlation ($R^2 = 0.936$). Dams-Out higher flow months tended to correlate with Dams-In higher flow months, while Dams-Out lower flow months corresponded with Dams-In lower flow months.

A Pearson's coefficient of determination was found for the correlation between USGS streamflow and Dams-Out streamflow (Table 15). The analysis indicated a strong positive correlation ($R^2 = 0.898$). Simulated higher flow months correlated with observed higher flow months, and simulated lower flow months correlated with observed lower flow months.

CHAPTER V

DISCUSSION OF RESULTS

**Summary**

The first purpose of this research was to produce a realistic model of watershed conditions in the Grand River Watershed, using SWAT. Statistics produced from the dams-in scenario confirmed the difficulty in using SWAT to accurately simulate hydrological conditions in the delineated watershed. Pearson's coefficient of determination demonstrated strong agreement between calibrated/validated data and observed data, and between dams-in data and dams-out data. However, most error statistics indicated that the model-produced values were not acceptable for representing watershed conditions.

Despite the amount of error associated with the calibrated and validated data, results followed roughly the same pattern of streamflow throughout the year as the observed USGS streamflow. The cause of the overestimated results is unclear, but may be related to inaccurate representations of infiltration, with which SWAT has been known to have errors (Kleinschmidt, 2010). Southern Michigan has a relatively flat topography, and a DEM with a finer spatial resolution may have better depicted topographic variations and natural flow basins. Additionally, selecting a greater minimum size than 3572 hectares per HRU to depict the frequency and extent of streams would have reduced some of the stream network, but perhaps also reduced the average monthly streamflow. Errors may also be attributed to the bias of the NSE and $R^2$ statistics towards higher flow events (Arnold et al., 2012b), as individual seasons had more extreme NSE values which affected the annual NSE statistic.

The second purpose of this research was to determine if a significant difference existed between streamflow in a dams-in scenario and a dams-out scenario. This significance was contingent upon successful replication of baseline watershed conditions. Were the calibrated and validated results to fall within 10% of the observed results, results of the dams-out scenario would be more valid. While the increase in streamflow from a dams-in scenario to a dams-out scenario was rather high, it is not considered statistically significant because of the difference between calibrated/validated data and observed data.

The third purpose of this research was to make recommendations for mitigation and management of the dams and infrastructure in the study area of Lansing. With respect to the third purpose of this research, the City of Lansing should still consider flood mitigation and waterfront redevelopment options in association with dam removal and the potential for increased streamflow in the study area. Since city officials have already identified dam removal as the best management practice, I would recommend for an Environmental Impact Statement (EIS) to be completed by the city to determine the cumulative effects of removing both dams.

**Study Limitations**

Limited stream flow, land use/land cover, and weather data exists from prior to construction of the Moores Park Dam in 1908 and the current North

Lansing Dam in 1936. Thus, calibration and validation of the model was not assessed in a dam-out scenario. Simulation of stream flow conditions without the dam in place still occurred in this research, with the validity of the simulation determined by the accuracy of the calibration and validation under a "dam-in" scenario. Simulation of the dam-out scenario followed the same temporal span as in the dam-in calibration and validation procedure.

Simulating dams as reservoirs is dependent on the secondary data content provided by the National Inventory of Dams. As depicted in Table 3, the Inventory provides variables such as maximum storage, normal storage, maximum discharge, and surface area for dams registered in the database. The surface area provided for each dam was used to estimate the surface area when filled to the emergency spillway in hectares (RES_ESA). Similarly, RES_EVOL the volume to fill to the emergency spillway in$10^4$ $m^3$ (RES_EVOL) and RES_PVOL the volume to fill to the principal spillway in$10^4$ $m^3$ (RES_PVOL) were based off the maximum storage and normal storage, respectively. The surface area when filled to the principal spillway in hectares (RES_PSA) was inferred from RES_ESA using the ratio of RES_PVOL to RES_EVOL at each dam. The most notable obstacle in dam simulation was the lack of available monthly streamflow data at each dam. This limitation was remedied by substituting the maximum discharge at each dam for the target release flow in $m^3/s$ (RES_RR). Changing the RES_RR value at each dam affected model output by indicating that each dam stores and releases differing volumes of water.

Conducting watershed modeling was made difficult by the ongoing evolution of watershed conditions through time. Most waterways undergo significant change over the span of several decades, including changes to channel roughness, channel slope, magnitude of stream meandering, soil infiltration rates, degree of overland flow of precipitation, and land use allocation. Results generated for a modern time-scale should be carefully applied to past watershed conditions with respect to information obtained about the past conditions and uses of the watershed.

Age of the Moores Park Dam and the North Lansing Dam limited the ability to calibrate and validate the streamflow conditions before dam construction. Confidence regarding the simulated streamflow conditions without the dams in place was drawn from the degree of success in calibrating and validating modern-day streamflow conditions under the baseline environment in the study area.

SWAT has been known to have inaccuracies in producing statistically significant flow estimations. Three common error scenarios in hydraulic calibration include the model failing to simulate peak flow events, the model overpredicting surface flow and base flow throughout the year, and the model lagging observed flow despite following the pattern of observed data (Arnold et al., 2012b).

This research utilized manual hydrological calibration through adjustment of individual hydrologic parameters during the sensitivity analysis and subsequent comparison of modeled streamflow output to the observed streamflow output. Calibration and validation for streamflow should be process-based and account for hydrologic variables including evapotranspiration, surface runoff, groundwater recharge, lateral flow, and deep aquifer recharge (Arnold et al., 2012b). While several studies have utilized manual calibration techniques, there is also the option to use SWAT-CUP software (SWAT-CUP, 2017), which performs automatic calibration and validation of the SWAT output. The user has the option to use five different algorithms in SWAT-CUP to account for prediction uncertainty in the model (Arnold et al., 2012b).

To accurately depict the streamflow values for the watershed, I would have had to adjust hydrologic parameters beyond realistic values for the Grand River Watershed. Some trial calibrations produced a yearly streamflow amount within 10% of the observed yearly streamflow amount. However, this required adjusting GWQMN, the threshold water depth in the shallow aquifer required for the base flow to occur, to a high value (~1000mm). This parameter increase caused the baseflow index to fall below 5%. As mentioned in Chapter III, the baseflow index in the study area is typically between 50-70% (Santhi et al., 2008).

Manual calibration of SWAT was limited to hydrologic calibration and did not include sediment and water quality calibration. This decision was made since the research was mainly concerned with streamflow, and because of the difficulty in obtaining secondary data for sediment and water quality in the watershed. Bosch (2008) acknowledged from his study of the Huron and Raisin

River watersheds that SWAT can be improved with greater availability and collection of water quality data and sediment transport modeling.

Despite ongoing efforts to adjust hydrologic parameters and calibrate/validate baseline streamflow results, the model was unable to replicate conditions in the study area. Modeled streamflow was only able to statistically match observed streamflow with extreme adjustment of hydrologic parameters beyond the acceptable values for SWAT. However, from an urban planning perspective, the overestimated model results in both the dams-in and dams-out scenarios are still useful. If Lansing city officials were pitching the need to remove these dams and redevelop riverfront infrastructure, having model overestimations would highlight the potential of increased streamflow with the dams removed. This potential for increased streamflow may be most significant for peak flow events during late winter or early spring flooding, when greater deviations in streamflow from the long-term average would be expected.

CHAPTER VI

# CONCLUSION

## Final Thoughts

The Soil and Water Assessment Tool (SWAT) was utilized to study the effects of removing the Moores Park and North Lansing Dams on the streamflow characteristics of the Grand River. The purpose of this research was to model baseline watershed conditions, determine the difference in streamflow between a dams-in and dams-out scenario, and suggest waterfront mitigation and management in the study area.

While baseline conditions were modeled with statistical significance during individual seasons, collective yearly results were not accurate. Therefore, conclusions regarding the increase in streamflow between a dams-in and dams-out scenario may not reject the null hypothesis if the study were to be further calibrated for sediment and water quality.

Despite broad difficulties in producing a statistically significant model, Lansing city officials should still consider dam removal as the best mitigation measure for these aging structures. Further model calibration may demonstrate that the projected increase in streamflow in a dams-out scenario is viable and warrants proactive measures. Dam removal would likely necessitate fortifying levees along the Grand River, but would also reduce the risk of significant property damage and loss of life from structural failure.

## Research Contributions

Despite difficulty in accurately representing conditions in the Grand River Watershed, SWAT remains a versatile and practical software in hydrological modeling applications. SWAT, in conjunction with ArcGIS, could store and compute a large volume of raster and vector data from varying sources. The software is relatively user-friendly, and the SWAT Input/Output documentation (Arnold et al., 2012a) thoroughly outlines model components, variables, and file information.

This research demonstrates the ongoing need to improve hydrological modeling for heavily impounded watersheds. While the dams-out scenario predicted a sharp increase in mean monthly streamflow, the calibration/validation results were not statistically significant. However, this potential increase in streamflow may be confirmed if the City of Lansing or Michigan Department of Natural Resources continued SWAT calibration of the watershed with improved sediment, water quality, and reservoir data.

## Considerations for Future Research

Future research would likely expand both the spatial and temporal span of this project. Adjusting the delineated watershed to include more of the Grand River Watershed and more USGS stream gauges downstream of the study area may offer more outlets to compare modeled and observed data. Expanding the duration of simulation would alleviate any outliers in the precipitation and climate data. The selection of a 30-year modeling duration would normalize climate data towards long-term averages and offset unusual peak or low precipitation events.

Model calibration and validation would include sediment and water quality calibration, provided adequate data coverage could be obtained. It is likely that sediment and water quality parameterization would have an indirect effect on the hydrology of the watershed.

The most accurate SWAT modeling results typically utilize primary field data in conjunction with secondary data (Bosch, 2008). Long-term research of the Grand River Watershed would allow for a field season of collecting sediment, water quality, and streamflow data upstream and downstream of each dam location. Field data could ease model calibration and accurate simulation of individual reservoir parameters.

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