Remote sensing framework reveals riverscape connectivity fragmentation and fish passability in a forested landscape

by

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Abstract

Fragmentation of stream networks by anthropogenic structures such as road culverts can affect the health of a catchment by negatively affecting the ecosystem's biota, their movements, abundances, and species richness. The challenge for resource managers is the prohibitive costs of locating, evaluating, and remediating problem structures at landscape-scales. There is a need for a framework to perform a desktop, landscape-scale evaluation and prioritization process using existing data that allows managers to make cost and ecologically effective decisions. I present a framework using publicly available LiDAR and orthophotography to locate and identify road crossings and evaluate fragmentation and passability for various fish species at the landscape-scale. My approach provides a valuable and cost-effective means of identifying potential stream crossing issues for multiple management objectives, e.g., fish passage, and thus the approach is an important step in the development of prioritization tools for restoration decisions by resource managers.

DEDICATION

To my friends, family, and mentors I have met through my life that have given me the motivation to pursue a life of conservation.

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Chapter 1: Introduction and Overview

Introduction

Fragmentation of stream networks caused by anthropogenic barriers has become a major issue for fish passability and connectivity among habitats (e.g., Trombulak & Frissell 2000; Khan & Colbo 2008). This is due to the increase in the industrial forest complex and other resource extraction that has rapidly expanded the network of roads in more remote locations and the resultant increase in stream crossings and associated structures, e.g., culverts (Lance & Balmford 2013; Januchowski-Hartley et al. 2014). If not properly installed or maintained, these culverts can become barriers to fish passage (Zwirn 2002; Park et al. 2008). The fragmentation of stream systems affects diadromous and local population of fish through loss of habitat, isolation of populations, genetic loss, and local extirpation (Morita et al. 2009, Nislow et al. 2011). Fixing issues sounds easy, but at the landscape-scales of 100's of km^2 typical of industrial resource operations, there is a need for a user-friendly, desktop framework to analyze stream crossings at large-scales and produce a prioritization of remediation strategies to overcome the river network fragmentation.

Maintaining connectivity of riverine systems is vital to fish populations persistence and survival (Torterotot et al. 2014; Erkinaro et al. 2017). Connectivity is the connectedness of different stream habitats that are responsible for healthy fish populations (Fullerton et al. 2010; Erkinaro et al. 2017). Throughout a fish's life history multiple types of habitat can be utilized depending on life stages, migration routes, habitat loss, or seasonality (Levings et al. 1995; Kahler et al. 2001). The fragmentation of these riverine systems will create habitat patches (Naslund et al. 1993; Cote et al. 2009). These habitat patches differ from terrestrial habitats because fish must follow the river networks and thus just one barrier can significantly impact migrations and movements which can be vital to a population's persistence and survival (Erkinaro et al. 2017; Torterotot et al. 2014). Additionally, barriers can alter stream flow affecting nutrient flow (Stanley & Doyle 2003), reducing fish health and populations (Nislow et al. 2011), and can lead to reduced genetic integrity of a population (Torterotot et al. 2014).

When roads are constructed and culverts are installed, there is also an increase in sediment deposit into the stream (Furniss 1990; Ottburg & Blank 2015). High flow events at improperly installed culverts can accumulate sediment, making them impassable to some fish species (Furniss 1990; Ottburg & Blank 2015). More generally, increased suspended sediments impact fish health and behavior, affecting feeding, species richness, and spawning success (see for examples, Chapman et al. 2014; Ottburg & Blank 2015). Increased velocity in culverts can cause scouring downstream and the result can be a perched culvert i.e., a hanging culvert (Furniss 1990; Norman et al. 2009). The water depth in the culvert and the height in which the fish must jump can become a barrier depending on the species (Burford et al. 2009; Bouska & Paukert 2009).

The most widely used, in-stream barrier causing fragmentation of stream networks in forested landscapes are culverts (Khan & Colbo 2008; Park et al. 2008).

They are used due to their cost efficiency, but if they are installed improperly, they are likely to fail and cause fragmentation problems (Zwirn 2002; Park et al. 2007). These barriers can be modified or replaced to restore connectivity but are often expensive and exceed most organizations budget and resource capabilities (Gibson et al. 2005; Poplar-Jeffers et al. 2009).

The number of culverts across the industrialized forest and other resource extraction landscapes is beyond counting (Khan & Colbo 2008; Park et al. 2008). In addition, there are rarely inventories of culverts at catchment-scales with exceptions for private landowners (not public information) and some provincial databases. Similarly, there are no inventories of failed/failing culverts. Currently, there are little to no management agencies that conduct passability surveys on individual culverts, due to it being labor and time intensive (Love & Taylor 2003; Hendrickson et al. 2008). Emerging remote sensing technology (i.e., Light Detection and Ranging, LiDAR) has been used to locate and identify possible barriers within streams to overcome these challenges (Diebel 2014; Kroon & Phillips 2015).

Prioritization is the process in which managers use variables such as amount of inaccessible habitat and cost of replacement or repair of barriers to increase connectivity within the system (McKay et al. 2016). This method is commonly used in situations in which there is limited funding dollars and there is substantial remediation that needs to be accomplished. While it is argued that prioritization at a landscape-scale is problematic due typically to lack of data, it is valuable when there are thousands of stream crossings within a catchment (O'Hanley & Tomberlin 2005; Januchowski-Hartley et al. 2014). Determining passability of barriers can be problematic due to the number of variables that determine passability (Anderson et al. 2012). It is vital to frame the project based on species-specific restoration goals and what abiotic and biotic factors are most beneficial for that species (McKay et al. 2016). The ability to pass culverts with different slopes varies greatly between fish species and can alter what barriers are prioritized (Anderson et al. 2012).

Direct observation of culverts is a traditional method used in ideal situations where time and funds are not limited (Bowen et al. 2006). This is not often the case and not realistic when there could be potentially thousands of crossings within a catchment (Kemp & O'Hanley 2010).

Measuring and remediating barriers at a catchment-scale is the most effective management action for improving connectivity for fish (Roni et al. 2008; Januchowski-Hartley et al. 2014). Januchowski-Hartley et al. (2014) used a boosted regression tree approach, but the method still depends on fine-scale, onsite data. Diebel (2014) used publicly available stream and road networks to locate potential crossings, as well as using an existing culvert inventory to verify which sites are culverts. They then used LiDAR Digital Elevation Model (DEM) data to collect elevation values to derive slope that required no fine-scale data and could be reproduced in other catchments. With slope being one of the primary factors that influence water velocity, and thus passability in a culvert (Doehring et al. 2011), determining passability remotely at the landscape-scale is possible. There is a need to understand the scale of fragmentation at a landscape-scale to remediate barriers for the biggest impact and, for the lowest cost (Bourne et al. 2011).

My objectives were to use high resolution LiDAR and Geographic Information System (GIS) tools to establish a stream crossing inventory remotely and measure the extent of fragmentation at the landscape-scale, i.e., 3000 km^2. Chapter 2 focuses on building the model and field-verifying the high-resolution LiDAR DEM interpretations (model) of stream crossings. Chapter 3 applies the models and examines the predictions for prioritizing barrier removal, i.e., fixing culverts and providing fish passage, for six fish species based on the upstream kilometers added with the removed barrier. Chapter 4 summarizes and discusses the implications using the models for culvert inventory and remediation projects.

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Chapter 2: A semi-automatic workflow for identification and classification of road crossings using LiDAR-data and ortho-imagery

Abstract

Stream connectivity issues are often encountered throughout forested landscapes due to the intersection of streams and road. Over time and if not properly maintained, culverts can lead to stream network fragmentation and the loss of availability of fish habitat. I developed a landscape-scale model (~ 3,223 km^2) to detect road crossings of streams in the Restigouche River catchment, New Brunswick using a novel Light Detection and Ranging (LiDAR) and geographic information systems (GIS) method. Using a combination of contemporary hydrological modelling tools, I create a new and more accurate stream network compared to publicly available data that are currently used for regional management decisions. A combination of LiDAR DEMs and digital orthophotographs successfully classified stream crossings as culverts, bridges, fords, and drainage pipes. Culverts are the problem that most often disrupts stream connectivity. I digitized culverts using LiDAR data and calculated the length and slope of the culvert, from which I further predicted passability for Atlantic salmon (*Salmo salar*). My approach provides a valuable and cost-effective means of identifying potential stream crossing issues for multiple management objectives, e.g., fish passage, and thus, the approach is an important step in the development of prioritization tools for restoration decisions by resource managers.

Introduction

Connected, productive freshwater ecosystems are imperative for maintaining healthy fish populations. A loss of habitats or connectivity among habitats can impede Atlantic salmon (*Salmo salar*) migrations and movements which can be detrimental to a population's persistence and survival (Erkinaro et al. 2017; Torterotot et al. 2014). After hatching, juvenile Atlantic salmon spend their first 1-8 years in freshwater streams until smoltification, after which an ocean-ward migration ensues to bring the salmon to feeding grounds at sea (e.g., Metcalfe & Thorpe 1992). After 1-2 years at sea, salmon return to their natal freshwater systems to reproduce (e.g. Gibson 1993). Many of the spawning and productive habitats for juvenile salmon are found in upstream, smaller order (1^{st}-2^{nd}) tributaries of the catchment that are accessed via the corridors of the river network (Heggenes 1990; Erkinaro et al., 2017), and barriers restricting access to these critical habitats are known to have detrimental population-scale impacts (Mahlum et al. 2013; O'Hanley & Tomberlin 2005).

Connectivity for fish across riverscapes is a metric of the physical connectedness of habitats. Fragmentation of this network can alter fish populations by restricting material and nutrient flow (Stanley & Doyle 2003), reducing population size and survival (Nislow et al. 2011), and lead to reduced genetic integrity of a population (Torterotot et al. 2014). While species such as Atlantic salmon have adapted to the naturally changing connectivity of riverscapes, e.g., as altered by beavers (Scruton et al. 1998; Sigourney et al. 2006), landscapes where humans are rapidly constructing roads such as, the industrial forests of North America (La Marche & Lettenmaier 2001; Luce & Wemple 2001), can create serious connectivity issues for fish (Dudgeon et al. 2006; Fullerton et al. 2010).

The most common anthropogenic barrier causing fragmentation of river networks in forested landscapes are road crossings (Khan & Colbo 2008; Park et al. 2008). Many crossings, particularly in upstream tributaries, use culverts to facilitate the passage of water through road embankments (Clay 1995; Erkinaro et al. 2017). Culverts are cost-effective and ubiquitous in forested landscapes where densities can be up to two culverts/km^2 (Poplar-Jeffers et al. 2009; Januchowski-Hartley et al. 2014). When a culvert is installed, the stream is forced into a restricted pipe which alters the morphology of the stream channel (Furniss et al. 1990; Bouska & Paukert 2009). This modification affects stream discharge and sediment transport regimes (Wellman et al. 2000) leading to high velocity barriers (Love & Taylor 2003), decreased passability during low flows (Warren & Pardew 1998), sediment dams (Furniss et al. 1990), concentrated debris deposition (Furniss et al. 1990), and erosion of downstream pools that create impassable waterfalls for fish at the culvert outlet (Furniss et al. 1990). Culverts are problematic if installed without first considering passability of native fish species (Doehring et al. 2011); however, construction practices that overcome passage issues are improving (e.g., Hansen 2007; David & Hamer 2012; NBDE 2012; FPP 2015). Regardless of design and construction standards, culverts must be maintained because the lack of maintenance is often a leading cause of river fragmentation (Furniss et al. 1990; Park et al. 2008).

The fragmentation of streams by road culverts is a landscape-scale issue in heavily used landscapes most notably, industrial forests (Tinker et al. 1998, Park et al. 2008). The density of roads in an industrial forest can be as high as 0.47 km/km^2 (Maitland et al. 2015). Given the intensity of culvert use within these landscapes and high probability of anthropogenic fragmentation, the identification and prioritization of problematic culverts is a necessity for land and water managers. Identification and prioritization can be problematic where managing at the landscape-scale (O'Hanley & Tomberlin 2005; Anderson et al. 2012). Currently, most management agencies determine passability based on field surveys of individual culverts (Love & Taylor 2003; Hendrickson et al. 2008) which is both costly and labor intensive. Emerging remote sensing techniques may provide an efficient solution using the generation of maps delineating potential problem stream crossings to overcome these challenges (Diebel 2014; Kroon & Phillips 2015).

Once identified, prioritizing and fixing impassable culverts has operational benefits such as road safety but also restores access to habitats if the fix reestablishes fish passage. Despite uncertainties, prioritization is the most cost-effective way to approach a landscape-scale analysis (O'Hanley & Tomberlin 2005; Anderson et al. 2012). Barriers can also exist in series creating a cumulative negative impact, thus adding to the complexity of decision-making targeting increasing habitat (Branco et al. 2014; Kraft et al. 2019). This is complicated further due to uncertainty of the passability requirements for different fish species (Kemp & O'Hanley 2010; Kemp 2016). It is also recognized that barrier removal at a landscape-scale goes beyond a fish passage issue because improving stream network connectivity can create cascading ecological benefits (King & O'Hanley et al. 2016; Kraft et al. 2019).

Doehring et al. (2011) demonstrated that slope (also referred to as gradient) was a primary factor affecting passability through culverts for juvenile fish i.e., slope has a positive relationship with water velocity in the culvert (Bouska & Paukert 2010). Physical characteristics of the fish can limit passability of the fish based on swim speed, water velocity, and the length of time the fish can maintain that speed (Olsen & Tullis 2013; Khodier & Tullis 2018. Additionally, culverts with higher slopes can create "hanging" culverts at their downstream outlet that are a barrier if the fish cannot jump from the stream into the culvert (Taylor 2000; Burford et al. 2009). There have been several studies outlining how to measure fish passability for culverts (e.g., Rayamajhiet al. 2012; Khodier & Tullis 2018), including mathematical models (Kraft et al. 2019), Light Detection and Ranging (LiDAR) technology (Diebel 2014), and simulation software (Bourne et al. 2011).

Collectively, the aforementioned techniques have potential for creating a more detailed, landscape-scale inventory of culverts and fish passability. A complex framework was developed by Januchowski-Hartley et al. (2014) using 30m-DEM and boosted regression tree models to create fish passability index. I now have better resolution 1m-Digital Elevation Model (DEM) and improved Geographic Information

Systems (GIS) techniques that can improve the overall framework for effective applications for managers. I examined the validity of coalescing various GIS-based tools and LiDAR derived DEM's at the landscape-scale (approx. 3,000 km^2) to build and assess a new framework as a tool for locating, classifying, and evaluating stream crossings by roads. A high-resolution LiDAR model and a more precise road network provided by industry was compared with existing, publicly available road and stream networks and compared to determine improvement in detectability of stream crossings using the LiDAR data, and then to assess passability based on published criteria for several species. The framework I propose for landscape-scale analyses creates a more accurate, detailed inventory of stream crossings with an assessment of the culvert integrity, and it is a framework which is intuitive, effective, user-friendly, and cost-effective.

Methods

Study area

The Restigouche River has a total drainage area of ~13,000 km^2, 51% of which is located in New Brunswick and 49% in Quebec, Canada (Figure 2.1). The catchment lies in the Atlantic Maritime Ecozone and is underlain by calcareous bedrock and blanketed with glacially reworked surficial deposits (Rampton et al. 1984; Fyffe & Richard 2007). The catchment's geomorphology is characterized by steep valleys and its main stem geomorphology is predominantly a gentle relief (Amiro 1983). 96% of the land use is forestry (RRWMC 2015).

Hydrographic network delineation

LiDAR DEM data were obtained from the Province of New Brunswick's data online (Service New Brunswick-, SNB 2017) for 3,223 km^2 of the catchment acquired in the summer of 2016. The LiDAR mission produced a data set with a point cloud density of 6 points per m^2 with the root mean squared error (RMSE) = 6.3 cm and the vertical error was 12.4 cm at 95% accuracy (SNB 2017).

The LiDAR DEM was utilized to simulate surface water flow through the landscape (Lindsay & Dhun 2015). A known limitation with surface flow modeling is closed depressions, that is, cells within the DEM that do not have an outlet, which can affect the delineation of flow paths (Hayashi et al. 2003). Several methods have been developed to remedy these limitations (Tarboton 2015; Jackson 2013), of which I use a selective breaching method to reduce the number of false depressions in the DEM (Wall et al. 2015), i.e., this allows flow to pass through embankments by lowering the cells to the stream elevation (Wall et al. 2015). This method involves locating all stream-road crossings and lowering the elevation of the cells of the embankment surrounding the crossing to allow the simulated flow to pass through the road embankment, thus preventing a false, closed depression (Wall et al. 2015). The result was hydrologically corrected DEM that has all true and artificially derived flow paths with no closed depressions from man-made topography (Wall et al. 2015).

To extract the stream network, a flow accumulation threshold needs to be set in the DEM to determine the required number of accumulated hillslope pixels to initiate a channel (O'Callaghan & Mark 1984). A flow initiation threshold of 40 ha was chosen, i.e., how much of the upstream contributing drainage area is needed before a channelized stream is predicted to initiate. The threshold was determined by visually comparing different thresholds to the National Hydrographic Network and by examining boundaries in relation to permanent streams (SNB 2018). At 40 ha, I was capturing ephemeral streams. If the flow accumulation threshold was set to > 40ha, it has the potential to detect more stream crossings while accepting that false detections of the ephemeral streams was a possibility. My selective breaching model applied the "Hydrocutter" flow path model (Wall et al. 2015). I used "build statistics" to symbolize or stretch raster data to run geoprocessing data sets to a landscape-scale area in ArcGIS (ESRI 2018).

The New Brunswick hydrographic (3,867 km) and road network (3,673 km) were acquired from Service New Brunswick (SNB 2018; Table 2.1). The road network used for the LiDAR crossing model included logging and private roads that was acquisitioned from J.D. Irving, Limited (6,035 km; Table 2.1).

Culvert identification model development

The locations of stream crossings were identified using the "intersect" tool in ArcGIS, with the inputs being the road network and stream network and no buffer, resulting in stream crossing locations (ESRI 2018). The SNB data were used with the intersect tool to determine the locations of potential stream crossings identified only through public data. Then, the DEM derived stream network was analyzed with the private road network database which allowed for a comparison of crossings between public data and my models.

The process created multiple data layers: (1) a 1-m resolution DEM set to 35% transparency while using dynamic range adjustment; (2) a traditional hillshade created from the DEM with dynamic range adjustment; (3) a red-green-blue (RGB) filter on the DEM; (4) the 35% transparent DEM draped over the hillshade; (5) a 1-m resolution orthophotography layer (SNB 2018); (6) the road network; and (7) the stream network (Table 2.1). Various combination of these layers allows the user to help identify where the road crossed a stream, i.e., the layers can be interchanged to represent the crossing and emphasize topographical features. For example, if a crossing was thought to be a culvert, the 1-m resolution DEM with the hillshade was overlaid. This would make the channel and the embankment of the road more apparent. In some cases, culverts or other crossings can be identified directly from the 1-m orthophotography layer, if there are no trees or structure blocking the view (Diebel 2014). I field validated the results as in Diebel (2014; see below).

Roads can generally be identified with orthophotography. Otherwise, the road was identified on the DEM by a steep elevation gain, followed by no elevation change, then returning to a steep elevation decrease, which would be the embankments of the road (Figure 2.3). Culverts were identified by visually observing the stream channel becoming more restricted and passing the road on the DEM (Figure 2.3). If the culvert is large enough, it is apparent on the orthophotography and some culvert ends were visible on the DEM. Embankments were absent for bridges and there was no channel restriction.

For each stream crossing that was identified as a culvert, the culvert was digitized manually. This was accomplished by locating the lowest elevation of the up-and downstream ends of the culvert on the LiDAR DEM. Once points were created at the lowest point on each side of the crossing (up and downstream of the road embankment) the culvert ends can be digitized, and thus, giving the distance (m) or culvert length and elevations above mean sea level (MSL) or culvert slope automatically. A positive slope was assumed to represent the water flowing from the upstream, down through the culvert (Wilson and Gallant 2000). To establish a digital pathway between the upstream and downstream points, I used ‘model builder’ (ESRI 2018) which finds the nearest points with the same unique identifier and produces a table pairing these two points. The model then creates points from the table and connects them to generate a predicted (digitized) culvert. Length and elevations of the paired points were identified and used in a raster calculator to determine the slope of each culvert (Table 2.4).

I validated my culvert location predictions against field-verified measurements using 242 randomly selected, stream crossings proportionally distributed among Strahler stream orders 1 to 6 ensuring a minimum 10% of sites within each stream order. In the field, each crossing was classified as a culvert (regardless of culvert type or design), a ford (travel through the stream channel with no culvert), or a bridge. A false detection was a model prediction of a crossing that was either not present or where there was no evidence of water flow or a road-crossing structure in the field. Where there was no water flow, but a culvert was present I classified the site as a “culvert, no channel”.

At each site, the GPS coordinates were taken at the center of the road at the stream crossing using a Garmin GPSMAP 78s (accuracy of 3.6 m). The slope was measured with a Zip Level which measured the elevation change between the upstream and downstream ends of the culvert with an accuracy of 0.01 mm (Technidea 2012). For each culvert examined in the field, I compared the model’s and actual length and slope using a paired t-test ($\alpha = 0.05$).

Results

Provincial data and LiDAR crossing model comparison

The Hydrocutter model produced a stream network of 4,122 km which was an 6% increase from the provincial data set (3,867 km). The inclusion of private roads (industry data) increased linear road kilometers by 64% compared to the provincial road network. The Provincial hydrographic network and Provincial road network identified 431 potential stream crossings (Table 2.2). The LiDAR crossing model initially created 1,714 potential stream crossings before removing overlapping or duplicate points and the final model had 1,633 potential stream crossings: 1,182 (72%) first order, 285 (17%) second order, 122 (7%) third order, 38 (2%) fourth order, 2 (1%) fifth order, and 4 (1%) sixth order (Table 2.2). The total number of potential crossings increased by 278% using my improved road network and LiDAR derived stream network. The crossing density increased by 254% using my framework due to the increase in linear road kilometers and linear stream kilometers.

I surveyed 242 sites that were: culverts = 78 (32%); drainage ditches = 45 (19%); bridges = 58 (24%); sites with no channel upstream = 25 (24%); a ford = 8 (3%); and false (desktop) detections = 28 (12%; Table 2.3). The accuracy of the model predictions ranged from 90-100% (Table 2.3). A false detection of a stream crossing occurred where a stream and road did not cross, which was the result of incorrect flowlines or geometry of the roads in these two networks, respectively. These false detections occurred most frequently in first order streams.

Culvert analysis

A total of 78 culverts were compared with field data. The predicted culvert lengths ranged from 5 m to 74 m and slopes ranged from 0.03% to 4% (Table 2.5). Field and LiDAR predicted culvert lengths showed no statistical difference (Table 2.5), as were field and LiDAR predicted slopes (Table 2.5).

Discussion

My GIS framework for identifying culverts at a landscape-scale proved successful both in terms of increasing identification rate for culverts (278%) and accuracy of culvert characteristics derived from LiDAR (length and slope). I correctly identified the type of stream-road crossings at over >90% of any stream order and at 100% at Orders >2. My framework demonstrates that a large spatial and remote landscape can be assessed quickly and accurately at the desktop.

Uncertainty exists in my approach. There were false detections and model-derived sites with no channel upstream occurrences in first and second order streams (15%). These results could stem from the DEM, its hydrological conditioning, and the flow initiation threshold chosen to derive surface flow channels for my stream network. Wall et al. (2015) demonstrated that flow accumulation can greatly alter the sensitivity in the Hydrocutter model if mapping ephemeral streams or closed depressions in the landscape. I chose a high flow accumulation threshold (40 ha) to derive my surface flows, as compared to the National Hydrographic Network and this resulted in an increased number of crossings in my DEM. Overall, my approach led to false detections in small, 1-2 order streams only and this should be considered by managers, where such smaller catchments are more relevant to their stream network.

Culvert passability is mostly governed by stream discharge and in-pipe velocity, and these characteristics can change over the life of a culvert (Warren and Pardew 1998; Gibson et al. 2005). Slope may be a useful analogue to determine passability (Anderson et al. 2012; Bourne et al. 2011). Culvert slope has been used to determine passability in previous studies (MacPherson et al. 2012; Januchowski-Hartley et al. 2014). Bourne et al., (2011) using a hydraulic modeling system, FishXing, calculated that maximum slope of 4% will allow passage of adult Atlantic salmon, a species that is prevalent throughout the Restigouche River. Provincial regulations in New Brunswick require culverts to be installed with a maximum slope = 0.5% without necessitating installation of baffled culverts. For framework-identified and field visited culverts, 75% exceeded this culvert slope guideline in the assessed Restigouche River catchment, and <5% exceeded the 4% maximum slope that could restrict adult salmon passage (Bourne et al. 2011). Such passability criteria are considered uncertain for many species (Cote et al. 2009; Mahlum et al. 2013), but my framework demonstrates how quickly any species-specific, landscape-scale assessment could be addressed.

Conclusion

The goal of this research was to present a novel framework and model for locating, identifying and analyzing stream crossings remotely at a landscape-scale. I correctly identified the type of crossing visually from LiDAR DEM and orthophotography with >90% accuracy and at 100% for stream orders > 2. Additionally, the method of finding the lowest cell on each side of the embankment was a repeatable process and produced an accurate measurement of culvert slope and length. As a result, the model was well suited for a landscape-scale analysis to remotely locate, identify, and evaluate stream crossings, and to be cost-effective if the LiDAR data is available. As LiDAR technology continues to advance and such data becomes more readily available, land managers and conservation organizations can be equipped with quick, efficient, and accurate landscape-scale tools for many management decisions such as my stream connectivity analysis. As one example, 75% of the culverts in my study area exceeded the Provincial guidelines for non-baffled, closed-bottom culvert slope which suggests policy and practice are not in sync. The results facilitate a prioritization exercise for stream crossings related to road maintenance, fish passage, and restoration efforts and which I will explore in a future study.

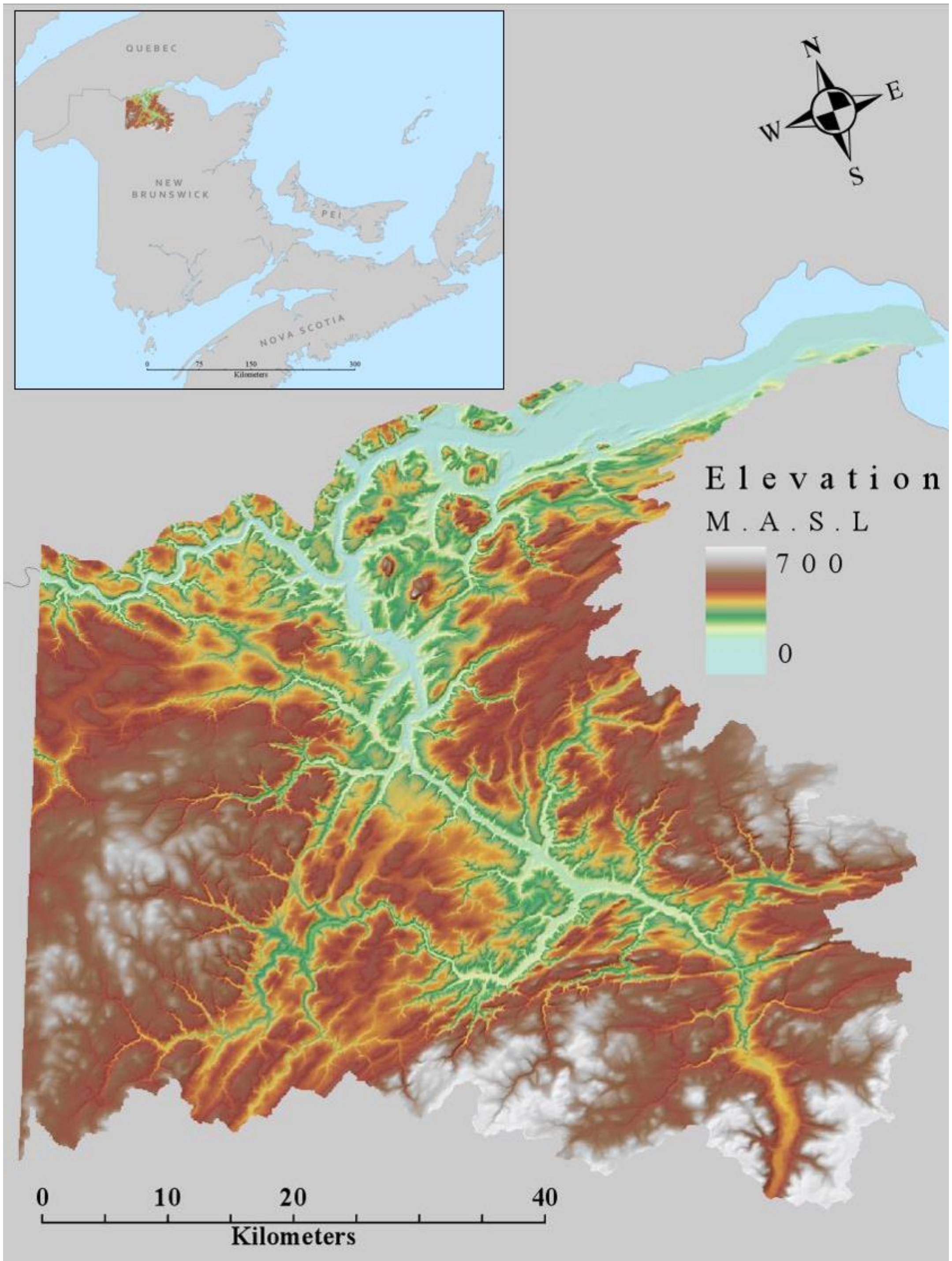

Figure 2.1. LiDAR range and sample area of the Eastern New Brunswick portion of the Restigouche catchment, New Brunswick, Canada.

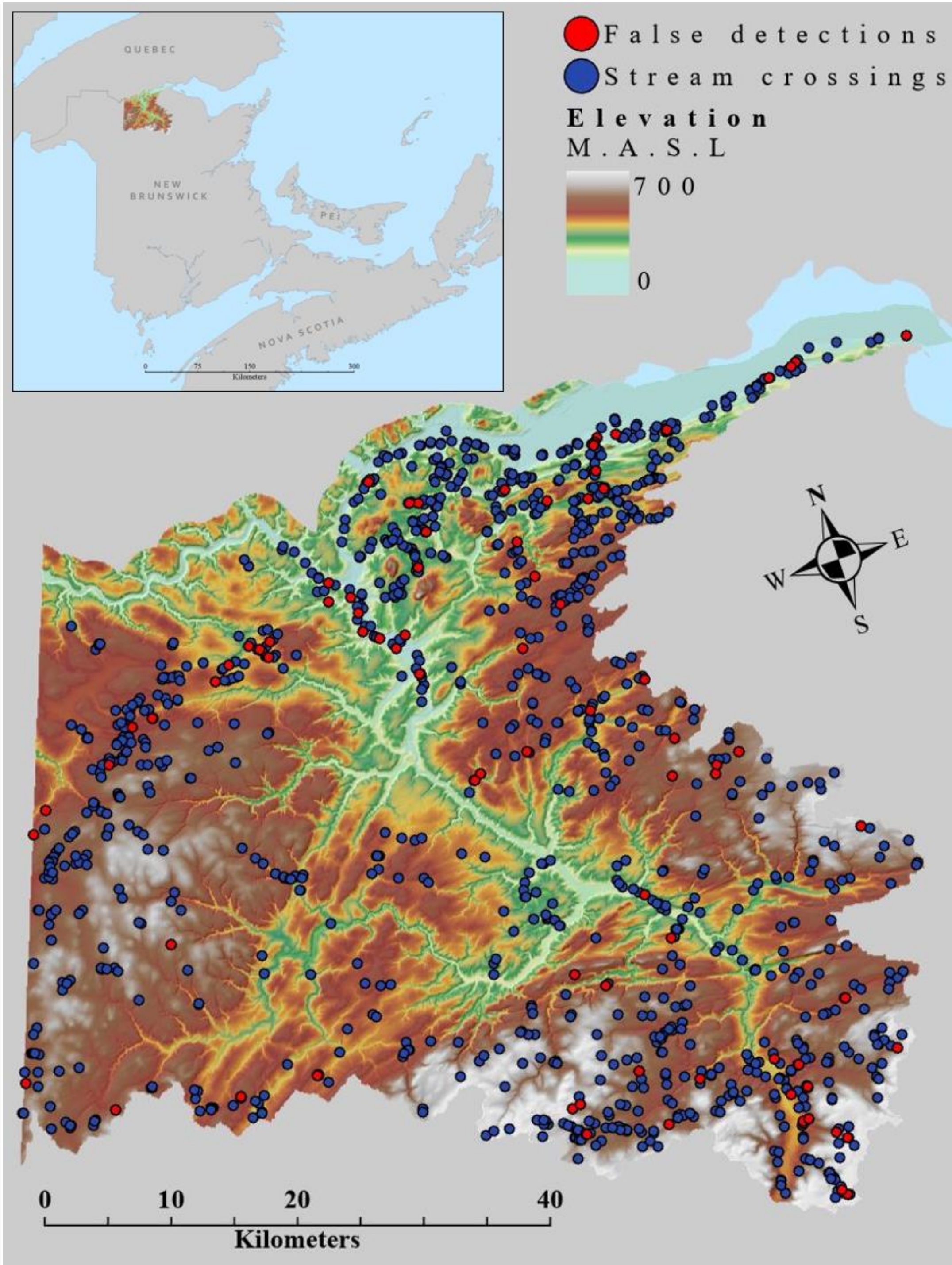

Figure 2.2. Stream-road crossings and false detections based on the results of the LiDAR crossings model in the Restigouche catchment, New Brunswick, Canada.

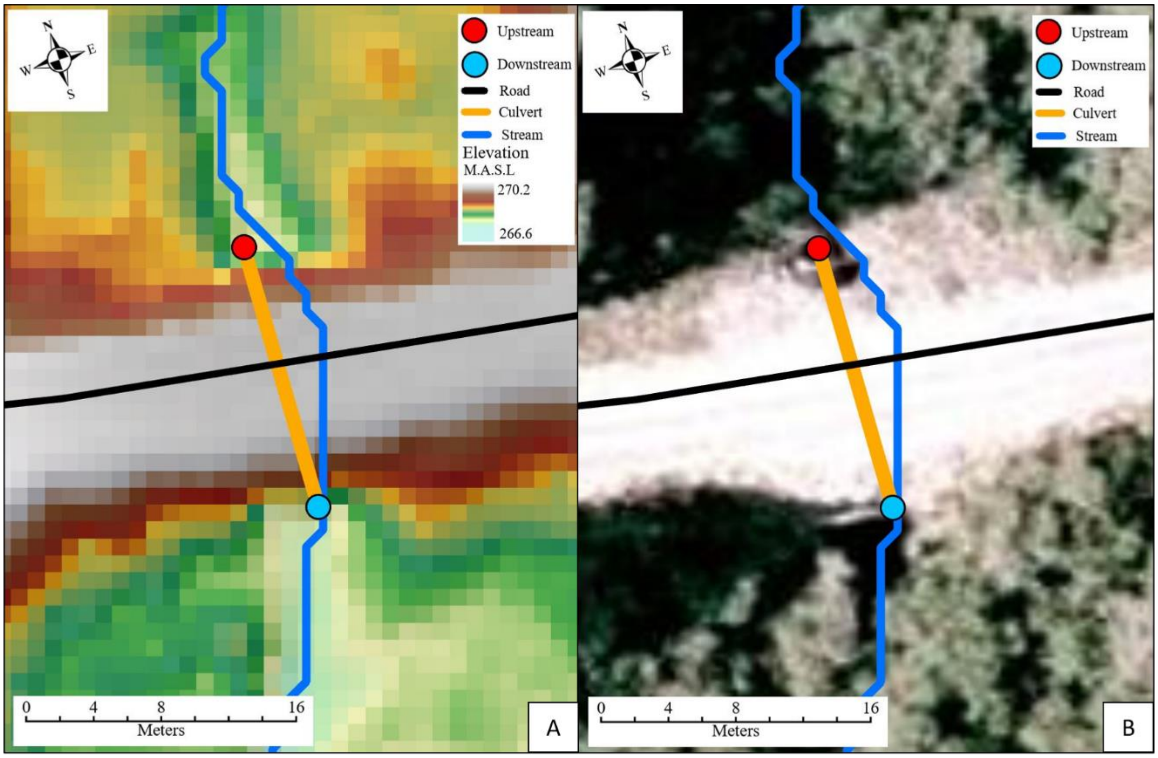

Figure 2.3. (A) Digitized culvert using LiDAR crossing model overlaid with a hillshade DEM, and (B) orthophoto reference showing digitized culvert.

Table 2.1. Overview of variables used for stream crossing framework, data type, and source.

Variable	**Data type**	**Source**
LiDAR DEM	Raster	Service New Brunswick
Provincial road	Feature	Service New Brunswick
J.D. Irving, Limited road network	Feature	J.D. Irving, Limited
Provincial stream	Feature	Service New Brunswick
Modeled stream	Raster Derived (Flow accumulation threshold 40 ha)	DEM Hydrocutter (Wall et al., 2015)

Table 2.2. Number of stream crossings by Strahler stream order based on provincial data (SNB, 2018) and the LiDAR modeled data.

Stream order	Provincial data n = 431	LiDAR crossing model n = 1633	Detection difference (%)
1	181	1,182	553
2	127	285	124
3	82	122	49
4	35	38	9
5	2	2	0
6	4	4	0

Table 2.3. Sites survey in the field (n = 242) by Strahler stream order and model accuracy (%).

Strahler Stream Order	Culvert (model accuracy)	Drainage Ditch (model accuracy)	Bridge (model accuracy)	No channel upstream (model accuracy)	Ford (model accuracy)	False Detection
Order 1	30 (93%)	37 (100%)	4 (100%)	20 (90%)	7 (100%)	22
Order 2	30 (93%)	3 (100%)	0	4 (100%)	1 (100%)	2
Order 3	18 (100%)	3 (100%)	15 (100%)	1 (100%)	0	3
Order 4	0	2 (100%)	33 (100%)	0	0	1
Order 5	0	0	2 (100%)	0	0	0
Order 6	0	0	4 (100%)	0	0	0

Table 2.4. Workflow for LiDAR stream crossing model including digitizing and extracting culvert slope from the DEM.

Step	Workflow Process
1	Process LiDAR to create DEM (create DEM from LiDAR point cloud data)
2	Use the "Intersect" tool using the stream network and road network to find all potential stream crossings
3	Use the LiDAR DEM, Orthophotography, or a combination to identify the crossing type (culvert, bridge, etc.)
4	Crossing data #1: Mark crossing point (culvert end) as lowest elevation on upstream side of crossing embankment. Repeat marking lowest elevation for downstream side
5	Crossing data #2: Extract the X, Y (geospatial) and Z (elevation) for crossing (culvert) down- and upstream points
6	Compute crossing/culvert length and slope
7	Run diagnostics on individual crossing/culvert as desired: For example, "if slope <0.5%, then passable."

Table **2.5**. Mean ± 1 SD of culvert length and slope measured using the remotely collected LiDAR DEM framework compared to field observations. With a T-test and p-values comparing the two.

	Length				Slope			
Stream Order	Mean LiDAR	Mean Observed	T-Stat	p	Mean LiDAR	Mean Observed	T-Stat	p
1	12.4 m	12.4 m	-0.26	0.79	0.93%	0.85%	-0.59	0.56
2	13.5 m	13 m	-0.24	0.81	1%	1%	0.08	0.93
3	22.6 m	21.4 m	-0.1	0.92	1.20%	1.20%	-0.26	0.8

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Chapter 3: A framework to assess landscape-scale fragmentation of stream networks and fish passability for resource managers

Abstract

Fragmentation of stream networks from anthropogenic structures such as road culverts can affect the health of a catchment by negatively affecting the ecosystem's biota, their movements, abundance, and species richness. The challenge for resource managers is the prohibitive costs of locating, evaluating, and remediating problematic structures at landscape-scales. There is a need for a framework to perform a landscape-scale evaluation and prioritization process using existing data that allows managers to make cost and ecologically effective decisions. I present a framework using publicly available LiDAR and orthophotography to locate and identify road crossings and evaluate fragmentation and passability at the landscape-scale. My results for a 3,223 km^2 area identified 1,052 stream crossings of which, 32% were culverts, and 12% of the total stream network was potentially blocked by these culverts. The 10 culverts restricting the most stream kilometers restricted >34% of the potential stream habitats for six species of fish. With this framework, managers equipped with imagery can create a comprehensive stream crossing database with only minimal funding, make an effective assessment of instream barriers, and prioritize removals at a landscape-scale, thus providing an effective assessment tool in their habitat restoration process.

Keywords

stream crossing, culvert, LiDAR, orthography, GIS, ecological connectivity,

Introduction

Anthropogenic, in-stream structures such as dams, weirs, and road-crossing culverts are widespread barriers causing significant fragmentation of catchment ecosystems (Trombulak & Frissell 2001; Anderson et al. 2012). Effects of fragmentation are emerging as an important issue for managers and planners because in-stream barriers can impact ecosystem structure and function, i.e., the stream ecosystem's goods and services (e.g., Mahlum et al. 2013; Torterotot et al. 2014; Erkinaro et al. 2017).

Culverts are a hydraulic structure that carry water under roadways or other embankments (Clay 1995; Erikinaro et al. 2017). Pipe-culverts are the most common in-stream structure used at stream crossings due to their low costs (Khan & Colbo 2008; Park et al. 2008). They can be variable in size, e.g., as small as 30cm in diameter, and most often installed under a roadway such that road traffic is unimpeded at the crossing (e.g., NBDE 2012). Culverts are typically installed for the singular purpose of transporting water through stream crossings and thus without consideration of fish passage (Blakey et al. 2006; Makrakis et al. 2012). While there are guidelines intended reduce environmental impacts on fish (e.g., Chilibeck 1992; NBDE 2012), culverts alter stream morphology and thus passability (Poplar-Jeffers et al. 2009; Price et al. 2010).

Culverts often are barriers for fish passage because they are not natural pathways and alter fish behaviour and movement patterns (Fahrig 2003; Gibson et al. 2005). The passability potential drops again when culverts are poorly installed or not maintained.

More broadly, problem culverts can disrupt aquatic ecosystem connectivity or the 'exchange pathway of matter, energy, and organisms' (Ward & Stanford 1995). The fragmentation of natural waterways creates habitat patches (e.g., Frissell et al. 1986; Cote et al. 2009) that can threaten habitat availability, biodiversity, and abundance (e.g., Calabrese & Fagan 2004; Khan & Colbo 2008; Nislow et al. 2011). Problem culverts can be repaired or removed to re-establish connectivity, but often there is competition for limited restoration funding and culvert repairs which are necessarily expensive become a low management priority (Gibson et al. 2005; Poplar-Jeffers et al. 2009). There is a growing desire to identify and establish a priority for remediating these culverts to improve ecological connectivity across catchments (Januchowski-Hartley et al. 2013; Erkinaro et al. 2017).

Previous studies have demonstrated that the slope of the culvert, the local hillslope gradient, and the elevation drop at the culvert influence fish passability at stream crossings (e.g., Peake et al. 2008; Bourne et al 2011). With adequate funding and time, managers and planners could assess and evaluate the passability of stream crossing (Bowen et al. 2006); however, this is not a feasible for the increasingly important, landscape-scale planning, e.g., forest management, where thousands of existing stream crossings need to be evaluated, each with variable access (Kemp & O'Hanley 2010). Potential assessment parameters are now measurable with remotely sensed data (e.g., Diebel 2014; Januchowski-Hartley et al. 2014). Januchowski-Hartley et al. (2014) demonstrated that statistical models can be used at the landscape-scale to predict passability of culverts using boosted regression trees if there is fine-scale data collection to build the models. Diebel (2014) demonstrated that culvert slope can be extracted from Light Detection and Ranging (LiDAR) digital elevation models (DEM). These evolving technologies and interpretations demonstrate the promise of measuring passability of culverts and therefore stream connectivity remotely. The challenge is moving from fine-scale to a landscape-scale, both technically and cost effectively (Kemp & O'Hanley 2010).

The goal of this research was to develop a framework and practical GIS tool to assess stream network connectivity and fish passability across large landscapes using remote-sensing data, specifically LiDAR generated, digital elevation models and orthophotography. I identified culverts and measured attributes predicted to impact passability, that is, slope (Kemp & O'Hanley 2010), elevation drop (Diebel 2014), and stream gradient (Walters et al. 2003). Based on existing literature, I assessed passability for six stream fishes including examining the number of potential barriers and stream kilometers restricted. Based on my analyses, I produce an assessment of cumulative effects of barriers removed to demonstrate how the framework and tool could be applied by managers and planners.

Methods

Study area

The Restigouche River catchment lies along the New Brunswick-Quebec border (Figure 3.1). The study site encompassed ~ 3,200 km^2 mostly within New Brunswick (Figure

3.1). The catchment is located within the Atlantic Maritime Ecozone and is blanked with glacially reworked, surficial deposits (Rampton et al. 1984; Fyffe & Richard 2007). The sediment is mainly composed of sandstone, shale, and limestone making it a highly dynamic river vulnerable to erosion (RRWMC 2015).

Data

A full description of the GIS workflow is provided in Chapter 2, and only a generalization of the process follows herein. Light imaging, detection, and ranging (LiDAR) was used to create a digital elevation model (DEM) which was available along with 1-m RGB orthophotography from the Province of New Brunswick (Service New Brunswick-SNB 2017). The 3,223 km^2 of LiDAR was collected from June to October 2016. Airborne LiDAR data, in LiDAR Data Exchange File (LAS) format, had an average point cloud density of 6 points/m^2, root mean squared error (RMSE) = 6.3 cm, and a vertical error = 12.4 cm at 95% accuracy (SNB 2017).

A stream network was created in ArcGIS using 1-m LiDAR DEM with a selective breaching model (Wall et al. 2015) to simulate flow through road embankments to simulate real life (Wall et al. 2015). A minimum upstream contributing area (flow accumulation) threshold of 40 ha was used to delineate a predicted channel network, i.e., this minimum flow initiation threshold defined the formation of a channel on the landscape.

The road network was generated in 2017 and provided by J.D. Irving, Limited (6,035 km of total roadways). Stream crossings were determined by using the 'Intersect' tool from ArcGIS which placed a point where the stream network intersected the road network (ESRI 2018). I used the "Delete Identical" tool in ArcGIS to delete all points except one within a 5m radius of each crossing.

I used multiple data layers: (1) a 1-m resolution DEM set to 35% transparency while using dynamic range adjustment; (2) a traditional hillshade created from the DEM with dynamic range adjustment; (3) a red-green-blue (RGB) filter on the DEM; (4) the 35% transparent DEM draped over the hillshade; and (5) a 1-m resolution orthophotography layer (SNB 2018); (6) road network; and (7) stream network. Various combination of these layers allows the user to help identify where the road crossed a stream, i.e., the layers can be interchanged to represent the crossing and emphasize topographical features. For example, if I thought a crossing was a culvert, I would overlay the 1-m resolution DEM with the hillshade. This would make the channel and the embankment of the road more apparent. In some cases, culverts or other crossings can be identified directly from the 1-m orthophotography layer, if there are no trees or structure blocking the view. These stream crossing classification methods are >91% accurate (Diebel 2014).

Among the layer combination, the most effective was the 1-m resolution DEM set to 35% transparency and the traditional hillshade created from the DEM, which visualized 781 (74%) of crossings. Other useful combinations were the previous two layers and the 1-m resolution orthophotography layer, which visualized an additional 221 (21%). Each stream crossing was classified as a culvert, ford, drainage pipe, no channel upstream, bridge, false detection, or DEM error (Figure 3.1). A culvert was identified by a stream network existing on both sides of an embankment (the road). A ford was where the stream crossed the road, but no in-stream flow related structure was present. A drainage pipe is a culvert that does not connect to any stream and typically, such run parallel to the road. No channel upstream is where there is a culvert or crossing present but there is no evidence of water flow upstream of the crossing. A false detection was a site where a road and stream do not exist or do not cross. A DEM error occurred where I identified by eye via the imagery, evidence of a stream and road crossing, but these features are not detected in my layers.

Analysis of Culvert Crossings

Culvert slope could be used a surrogate for fish passability. Doehring et al. (2011) demonstrated that slope was a primary factor affecting passability through culverts for juvenile fish i.e., slope has a strong positive relationship with water velocity in the culvert (Bouska & Paukert 2010). Physical characteristics of the fish can limit passability of the fish based on swim speed, water velocity, and the length of time the fish can maintain that speed (Olsen & Tullis 2013; Khodier & Tullis 2018). Additionally, culverts with higher slopes can create “hanging” culverts at their downstream outlet that are a barrier if the fish cannot jump from the stream into the culvert (Taylor 2000; Burford et al. 2009).

Culvert slope has been used as a metric for fish passability by several previous studies (Burford et al. 2009; Bourne et al. 2011; MacPherson et al. 2012; David et al. 2014; Khodier & Tullis 2018; Table 3.1). For each stream crossing that was identified as a culvert, the culvert was digitized manually. This was accomplished by locating the lowest elevation of the up and downstream ends of the culvert on the LiDAR DEM. Once points were created at the lowest point on each side of the crossing (up and downstream of the road embankment) the culvert ends can be digitized, and thus, giving the distance (m) or culvert length and elevations above mean sea level (MSL) or culvert slope automatically. A positive slope was assumed to represent the water flowing from the upstream, down through the culvert (Wilson and Gallant 2000). An elevation drop of >30cm at the culvert's downstream exit was also considered a potential barrier for all species based on existing literature, e.g., Januchowski-Hartley et al. (2013) and Diebel (2014).

I selected six common stream fishes to examine passability. These fish were Burbot (*Lota lota),* Lake Chub (*Couesius plumbeus*), Rainbow trout (*Oncorhynchus mykiss*), Atlantic salmon (*Salmo salar*), Brook trout (*Salvelinus fontinalis*), and Brown trout (*Salmo trutta*). To evaluate the passable conditions for each of the fish species, slope thresholds were collected from previous studies (Table 3.2). To assess the potential to add fish habitat with the removal of a potential barrier, I assumed that making an "impassable" culvert passable, i.e., removing it as a "barrier", then the upstream network (km) represented added habitat. This assumption doesn't consider the species-specific habitats upstream, but it provides a first step towards detailed assessments that managers can later develop. Beginning at the first “barrier” in a stream network, I calculated the stream distance (km) added between the barrier and the next barrier or multiple barriers when the stream network split into lower order tributaries. I next assessed the stream slope between barriers by examining slope at 100 m stream segments using the “Editor” and “Add Surface Information Tool” in ArcGIS. The maximum slope between “barriers” was compared against species’ gradient tolerances (see Table 3.1), and either classified as “barrier” or “added habitat”. Known waterfalls were added to the data set and I considered these to be impassable.

Prioritization

Prioritizing barrier removal begins with a management objective, e.g., increase available, potential habitat for fish species. Tools such as the Barrier Assessment Tool (BAT, The Nature Conservancy 2010) have been useful in quantifying the amount (distance) of stream that could be restored by removing a barrier of concern. Herein, I continued with an assumption that allowing access to upstream reaches was equivalent to adding habitat for fish, i.e., km of stream added. To prioritize among potential barriers to remove, i.e., considered a barrier because of culvert slope, culvert drop, upstream slope, or waterfall, I ranked each barrier based on the total km that would be potential available if the barrier were overcome (e.g., repaired). I report the expected improvement in cumulative passability for each species if the 10 culverts that restricted the greatest cumulative stream distance were made passable (Table 3.2).

Results

The landscape analysis covered an area of 3,223 km^2 and 4,100 km of streams. Full details are provided in Chapter 2. In summary, there were 1,052 crossings and 339 (32%) were culverts. Culvert slope varied from 0.15 – 16.38%. Forty-one percent of culverts had a slope < 2%, 24% had slopes 2-4%, 22% had slopes 4-8%, 13% had slopes >8%. The slope threshold for species was exceeded at 95-199 culverts (Table 3.1). Elevation drop was a static number (>30 cm) and the number of such barriers was 172 (for all species – Table 3.1). Most barriers (due to slope and drop collectively) were in first order streams (148 or 44%). There were few stream segments with multiple barriers (16 barriers or 5%).

Based on the assumption that removing a barrier added potential habitat where stream slope upstream or waterfalls were not impediments to movement, overcoming a barrier added a potential 431 to 503 km of habitat for Burbot, 431 to 503 km for Lake Chub, 426 to 431 km for juvenile Rainbow Trout, 285 to 431 km for Atlantic salmon, 237 to 431 km for Brook Trout, and 222 to 431 km for Brown Trout (Table 3.2). The 10 culverts that restricted the greatest length of the stream network could, if corrected, potentially add 123 km (25%) for Burbot, 123 km (25%) for Lake Chub, 123 km (29%) for juvenile Rainbow Trout, 85 km (30%) for adult Atlantic salmon, 83 km (35%) for Brook Trout, and 83 km (38%) for Brown Trout (Table 3.2). For example, fixing 10 culverts (5% of total barriers) increased potential habitats for Lake Chub by 125 km or 25%.

Discussion

The goal of this research was to develop a framework and practical GIS tool to assess fish passability across large landscapes using remote-sensing data, specifically orthophotography and LiDAR generated, digital elevation models. My framework identifies multiple crossing types and assesses culvert slope (Kemp & O'Hanley 2010) and outflow drop (Diebel 2014) as potential barriers. The framework then assesses the stream network upstream of the potential barrier and uses stream slope and presence of additional culvert and waterfall barriers to predict the potential habitat that may become available for different fish species. I demonstrated the feasibility of the framework for a >4,000 km^2 region with a >1,000 km stream network.

The framework is designed for landscape-scale analyses using remotely collected data, but there are inherent assumptions. First, LiDAR and other imagery is static data, and thus doesn't reflect temporal variability of the landscape, e.g., culvert repairs or additions over time or natural dynamic processes of riverine systems that alter habitats and accessibility over time (Fullerton et al. 2010; Ogren & Huckins 2015). Additionally, I and others define passability for fish at a culvert based on a culvert outlet drop and up- to downstream slope of the culvert as a surrogate for water velocity and therefore thresholds of swimming ability (e.g., Love and Taylor 2003; MacPherson et al. 2012). In part, this reflects the paucity of research assessing passability for fishes (e.g., Warren and Pardew 1998; Peake 2008; Mahlum et al. 2013; Gautreau 2017). Others have highlighted additional problems with my assumptions (Diebel et al., 2010; Kemp and O'Hanley 2010). For example, high velocity barriers can impede fish passage (Peake 2008; Goerig et al. 2015), but high velocity or culverts with outlet drops can be passed by fish under certain water conditions (Kemp & O'Hanley 2010; Anderson et al. 2012; Gautreau 2017). As passability thresholds improve with new studies, e.g., better slope thresholds and culvert design differences (internal baffles/smooth pipe/corrugated pipe, e.g., Bouska & Paukert 2010), the improved thresholds and criteria can quickly and easily insert into the framework to improve overall predictability for managers.

I attempted to identify natural barriers, specifically detecting waterfalls in the orthophotography and DEM, based on knickzones. A knickzone is a large change in elevation along a longitudinal gradient (Zahra et al. 2017). Zahra et al. (2017) created a tool to extract knickzones from DEM's. I applied the "knickzone extraction tool" on a 10m DEM and a curvature threshold value = 2 based on the terrain shape with moderate relief (Zahra et al. 2017). The results generated many false detections, (i.e., a waterfall was predicted, but no crossing feature was present when inspected visually). If the waterfall was identified visually, then the elevation drop can be used to determine if it's a barrier to fish passage. Zahra et al. (2017) explains that a significant range in stream gradients can cause an increase in detection of "false" barriers. My study area in the Restigouche River catchment ranges from 0- 700 MASL and thus the likely cause of the many false detections.

There are a number of caveats that were made when classifying what is and is not a barrier. An example of this is that there if the 30cm drop threshold was reduced, there would be an increase in the number of barriers. Additionally, while I can classify a culvert as passable, in reality, it could be clogged from debris and this is common due to lack of maintenance (Gautreau 2017). This indicates that there are more barriers present in the catchment than the findings of this study show. The goal of this study was not to find every culvert that isn't passable, but to identify the culverts that are barriers that are restricting the most habitat.

In general, it is extremely difficult to correctly characterize fish habitats in smaller streams at a landscape-scale using remote sensing methods that were available for this study. I used stream gradient as an indicator of potential habitat (Januchowski-Hartley et al. 2014), and then to demonstrate the framework's functionality for decision-making tools, examined stream length of acceptable gradients as a surrogate metric of habitat added by removing a barrier. It is a demonstration of a simple, desktop-derived metric to highlight the functionality and adaptability of the framework. The ability to assess actual aquatic habitats from remotely sensed images is improving (e.g., Praskievicz & Buege 2017; O'Sullivan et al. 2020) and I believe managers will eventually have on their desktops, both habitat and barrier maps at the landscape-scale derived from remotely sensed data that can be used effectively in decision-making.

Data Availability

The data that is used in the study, excluding the private JD Irving Ltd. road network, are openly available from Service New Brunswick at http://www.snb.ca/geonb1/e/index-E.asp

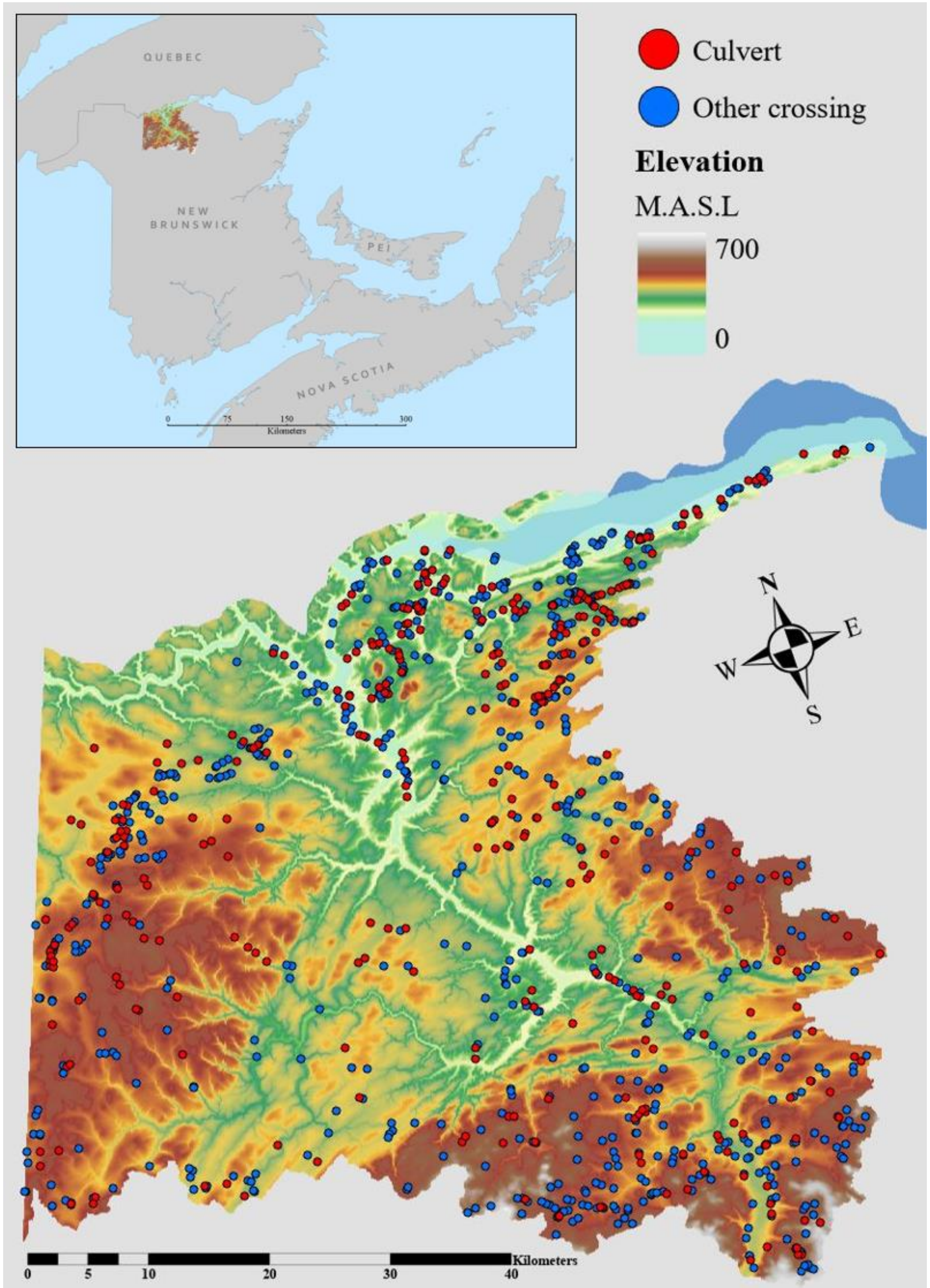

Figure 3.1. Culvert and other crossings (n = 1,052) generated by the LiDAR-based stream crossing modeling the Restigouche River catchment, New Brunswick, Canada.

Table 3.1. Thresholds for six species determining the slope when a culvert becomes impassable, the number of predicted barriers impeding each species, the kilometers of stream potentially restricted by the predicted barriers, and the estimated habitat (% of the stream kilometers) predicted to be inaccessible within the study area (see Figure 2.1).

Species	Slope threshold (%)	Number of barriers due to culvert slope	Number of barriers due to culvert elevation drop	Culvert slope restricted stream (km)	Culvert elevation drop restricted stream (km)
Burbot [1]	2	199 (12.2%)	172 (10.5%)	502.8	431
Lake Chub [2]	2	199 (12.2%)	172 (10.5%)	502.8	431
Juvenile Rainbow Trout [3]	2.5	173 (10.3%)	172 (10.5%)	425.7	431
Atlantic Salmon [4]	4	120 (6.9%)	172 (10.5%)	285.3	431
Brook Trout [5]	4.5	105 (5.8%)	172 (10.5%)	237.4	431
Brown Trout [6]	5	95 (5.4%)	172 (10.5%)	222.4	431

[1]MacPherson et al., 2012; [2]MacPherson et al., 2012; [3]David et al., 2014; [4]Bourne et al., 2011; [5]Burford et al., 2009; [6]Khodier & Tullis, 2018

Table 3.2. The amount of habitat, by species, restricted by 10 single culverts predicted to be barriers presented as stream kilometers that could be made available if the barrier was made passable, and as % of potential habitat added in the study area (~ 4,100 km).

	Top 10 culverts		
Species	**Stream restricted (km)**	**% of habitat restricted**	**% of barriers**
Burbot	123	25	5
Lake Chub	123	25	5
Juvenile Rainbow Trout	123	29	6
Atlantic Salmon	85	30	8
Brook Trout	83	35	10
Brown Trout	83	38	11

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Chapter 4: Summary

There is increasing expansion of natural resource development such as industrial forestry across Canada and around the world. Such activities continuously expand into remote locations, new roads are constantly built, and this means stream crossings continue to expand in number. This adds to the already unnumbered culverts that blanket landscapes across Canada. Managers of these resources and the altered natural river networks (ecosystems) understand and are searching for more detailed stream crossing inventories at landscape-scale.

With the increase of remotely assessed, data availability and quality (LiDAR, orthography), it is possible to identify and quantify fragmentation at a landscape-scale. Herein, I created a management tool based on models and a framework to locate, identify, and analyze stream crossings functionality at a landscape-scale at a 3000km^2. Using LiDAR and orthophotography and verifying with on the ground observations, the framework and models can discern types of stream crossings. The methods require GIS expertise, but the framework provides an easy and available method to measure fragmentation and prioritize individual barriers.

The models and framework presented demonstrate a desktop method in which managers who have limited funds but have access to existing LiDAR and orthography data can create an inventory of stream crossings and determine barriers at a catchment-scale. While I ground-truthed my barrier model results, there remain several caveats for this type of analysis. (1) LiDAR and Orthophotography are associated with a certain amount of resolution error and are only a snapshot in time. (2) There are more than the variables outlined in this study that determine passability (Anderson et al. 2012; McKay et al. 2016). Stream discharge, slope, and in-pipe water velocity can change temporally (Warren and Pardew 1998; Gibson et al. 2005). LiDAR data is collected during a singular moment and conditions may change after data collection. Especially if LiDAR data are several years old, assumptions have to be made regarding stream conditions being currently in the same condition as at the time of LiDAR data collection. The LiDAR was collected recently, in 2017, but that is not guaranteed for other LiDAR collected throughout the country. Accounting for that, I still believe that this is an effective first step for catchments to begin their remediation work. This is since slope has been determined a main component of passability (Anderson et al. 2012; Bourne et al. 2011).

While my results demonstrated the ability to identify stream crossings, false detections in Strahler stream order one and two were much higher than for larger streams and rivers. I believe that most of these false detections came from how the stream network was created from the LiDAR DEM. The stream derived from the LiDAR DEM was created with the assumption that the smaller order streams could be ephemeral streams (i.e., temporary streams, Wall et al. 2016). Data creation and preparation is a vital part that can alter the results of my crossings models based on the number of stream kilometers. For example, if I chose to not include possible ephemeral streams, I could fail to detect some smaller crossings in the head waters of my catchment.

I expected the LiDAR slope and field collected slope to be statistically significant based on Diebel (2014). While my results showed that the field and remote collected data did not have a statistical difference, my p-values varied from 0.79-0.93, 0.79 being stream order 1. I believe that this could be a function of smaller streams with smaller culverts, that could not be as defined as culverts that are in larger stream orders as demonstrated in Diebel (2014). While small streams did have the lowest p-value, the results are still strong and reflect real life.

Diebel (2014) was one of the first to utilize LiDAR in identifying crossings and measuring slope. I found expanding and refining the research allowed us to have a beneficial tool for identifying the extent of fragmentation. My Chapter 3 has not been published or available before. By framing all my research into a species-specific study, gave us valuable insight into how each species is affected by fragmentation.

Chapter 3 was built on Chapter 2 by expanding the culvert analysis to every culvert within my study site and prioritizing remediation efforts. Using my automated ArcGIS toolbox from Chapter 2, slope was extracted from each culvert. Using those slope thresholds, I applied specific fish species that were present in New Brunswick to measure the number of kilometers that were restricted by barriers. I highlighted the extent of fragmentation within the catchment by quantifying the number of kilometers restricted for each species. It is important to note that while I used stream gradient to identify stream reaches where some species couldn't access even if the barrier was remediated, it is with the assumption that they can't access the stream through all seasons. The caveat to this method is that there are multiple ways to prioritize barriers (i.e., ranking, scoring, budget) and can alter what is a high priority for remediation (Diebel 2014).

Final synopsis

As remotely sensed, landscape-scale data quality and coverage, becomes more available, the need for innovation and analysis of landscape-scale issues like barriers become more apparent (Januchowski- Hartley et al. 2014; Kemp and O'Hanley 2010). The quality and quantity of data is vital to making informed conservation management decisions.

As LiDAR becomes more readily available, there is a need for landscape-scale analysis to understand the extent of fragmentation within the catchment (Januchowski-Hartley et al. 2014. Understanding barriers and fragmentation at a large spatial-scale allows managers to make more informed decisions than fine-scale analysis (O'Hanley et al. 2014, Diebel 2014). As demonstrated in King and O'Hanley (2016, barrier prioritization at a large-scale proved to be far superior to any fine-scale analysis done previously. Often the issue with these landscape-scale analyses is the use of coarse-scale data and limited passability variables that can be collected (O'Hanley et al. 2013). Depending on the quality of data and assumptions made, the prioritization can drastically change when accounting for variables such as habitat quality, remediation costs, or other species-specific requirements. Often, there is several databases that are not in collaboration with each other or not publicly available. In these cases, there is a need to have one centralized database to achieve project and management goals (Kemp and O'Hanley 2010). In New Brunswick, for example, there is a centralized, public data hub collected by the Provincial government (Service New Brunswick) to compile GIS based data. To further this, there is a need for consideration on how to collaborate with various private and non-private organizations to combine data into one database to increase the quality and quantity of data available to the public. This is otherwise known as the findability, accessibility, interoperability, and reusability (FAIR) data. With this knowledge, managers should ensure that results found by remotely collected data should be field verified to ensure the correct decisions are made.

In conclusion, my thesis provides a valuable framework to create and analyze stream crossings and prioritize barriers for species at a landscape-scale. This thesis should be used as an initial analysis for managers that have LiDAR DEM available to create a detailed stream crossing inventory and prioritize based on slope for remediation of the barriers that are identified. Due to the success of this thesis, organizations such as the Gespe'gewaq Mi'gmaq Resource Council and the Restigouche River Watershed Management Council began targeting sites for remediation based on the framework presented in this thesis and will start restoring stream connectivity as early as Fall 2020.

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Arsenault, Michael. High resolution LiDAR and GIS model reveals extent of stream fragmentation across a forested landscape. Presented at the Atlantic Society of Fish and Wildlife Biologists (ASFB), 21 October 2018, Corner Brook, NL, Canada.

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