How effective have national parks in Canada been since their designation at preventing landscape fragmentation?

Clara Freeman-Cole

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By: Clara Freeman-Cole

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Signed by the final Examining Committee:

___________________________________ Chair & Examiner

Dr. Angela Kross

___________________________________ External Examiner

Dr. Chantel Markle

___________________________________ Supervisor

Dr. Jochen Jaeger

Approved by: ____

Dr. Craig Townsend, Chair of Department

____

Dr. Pascale Sicotte, Dean of Faculty

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ABSTRACT

How effective have national parks in Canada been since their designation at preventing landscape fragmentation?

Clara Freeman-Cole

Landscape connectivity has become an important focus of Canada's conservation goals. However, ongoing habitat loss and the break-up of habitat patches change a landscape's composition and configuration, creating additional barriers for wildlife movement and reducing its connectivity. Canadian national parks are at risk of becoming fragmented from direct and indirect stressors, and of becoming islands of natural habitat that are disconnected from their surrounding ecosystems. This study measured the landscape fragmentation of 43 Canadian national parks and paired control areas at key time-steps throughout their history, including before their designation to present-day. It also studied the divergence of fragmentation levels over time between the parks and their control areas. The Effective Mesh Size metric (m_{eff}) with both the CUT and CBC procedures was used to measure fragmentation levels, alongside a Progressive-Change Before-After Control-Impact Paired Series study design. The results demonstrate that overall, park protection across the Canadian National Park System has been somewhat successful in preventing landscape fragmentation within their boundaries, in comparison to unprotected control areas. Half of the parks and control areas had a very small change in fragmentation over time. In 35% of parks, fragmentation levels increased faster than in their control areas, and in 15% of parks, fragmentation levels increased more slowly than in their control areas. Older parks with a long history of human influence on the landscape are more fragmented than their control areas, whereas parks that have been designated more recently have a greater effective mesh size than their associated control areas. The regions with the most successful parks in terms of preventing fragmentation are the Taiga and Hudson. The variety of park sizes in these regions indicates that the size of a national park does not necessarily indicate its fragmentation levels. This study provides insights into trends of landscape fragmentation over a long time-period and relates them to different Parks Canada management strategies throughout the agency's history. The findings will help inform ecological connectivity programming and can indicate the effectiveness of Canada's efforts to achieve its conservation goals.

Keywords: Landscape fragmentation, Landscape connectivity, Protected Areas, Ecological integrity monitoring, Landscape change, Effective mesh size (m_{eff}), Progressive-Change Before-After Control-Impact Paired Series (PC-BACIPS)

Territorial Acknowledgement

I acknowledge that Concordia University is located on unceded Indigenous Lands. The Kanien'kehá:ka Nation is recognized as the custodians of the lands and waters on which Concordia stands. Tiohtià:ke/Montréal is historically known as a gathering place for many First Nations.

Mohkínstsis/Calgary is located on the ancestral lands of the Treaty 7 signatories, including the Siksika Nation, Piikani Nation, Kainai Nation, the Îethka Stoney Nakoda Nation (consisting of the Chiniki, Bearspaw, and Good Stoney Bands), and the people of the Tsuut'ina Nation. The Métis people of Alberta Region 3 also call Treaty 7 lands their home.

This work was undertaken in both Tiohtià:ke and Mohkínstsis.

It is crucial to highlight that the creation and maintenance of the national parks system in Canada was built on the forced removal of Indigenous peoples from their lands, under the prescribed ideals of "pristine wilderness". Inclusion of indigenous co-management and traditional indigenous use of now federally-governed land is not consistent across the National Park System, and so much of the system continues to exist in as a form of settler colonialism.

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For Mercy – sunshine personified during some of the gloomiest Montreal days.

Contributions of Authors

As the first author, I was responsible for the study conception, data collection and corrections, calculations, data analysis and interpretation, and the writing of the manuscript. The manuscript was co-authored by Dr. Jochen Jaeger, who advised on data calculations, analysis and interpretation, and who contributed revisions to the manuscript.

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List of Abbreviations

AkMM	Akami-Uapishk^{U}-KakKasuak-Mealy Mountains
BP	Bruce Peninsula
CBC	Cross-Boundary Connection [procedure]
CBH	Cape Breton Highlands
CUT	Cutting-Out [procedure]
m_{eff}	Effective mesh size
EIM	Ecological integrity monitoring
FG	Fragmentation geometry
IPCA	Indigenous Protected and Conserved Area
NPR	National Park Reserve
NPS	National Park System
OECMs	Other Effective Area-based Conservation Measures
PA	Protected Area
PC-BACIPS	Progressive Change Before-After Control-Impact Paired Series
PEI	Prince Edward Island
TDN	Thaidene Nëné

1- Introduction

Landscape fragmentation – the patchwork conversion and development of sites via perforation, incision, dissection, dissipation, shrinkage and attrition - has been increasing at a rapid pace in many parts of Canada, particularly due to anthropogenic land conversion for urban development, road construction, and agriculture (Cole et al., 2023). It is a “threat to landscape quality and the sustainability of human land-use”, affecting both wildlife and humans due to the impacts on wildlife habitats and ecosystem services (EEA & FOEN, 2011:52). Alongside landscape fragmentation, landscape connectivity – the degree to which landscape structure facilitates the flow of organisms through the landscape - is also a related fundamental concept of landscape ecology (Taylor et al., 2006). Landscape fragmentation reduces connectivity by creating barriers to animal movement to other natural areas, as well as creating overly small habitat patches for the species impeded, thus contributing to biodiversity loss (Moser et al., 2007; Girvetz et al., 2008). Both connectivity and fragmentation are essential to consider with regards to conservation efforts in Canadian National Parks and Park Reserves, since any increase in human activity in these areas is likely to affect them. They provide an indication of changes to the integrity of the native ecosystems of parks and their surrounding areas (Soverel et al., 2010). The Parks Canada Agency’s current official objective of protecting the national parks is to maintain and restore their ‘ecological integrity’ (Canadian National Parks Act, 2000; Fluker, 2010). Ecological integrity with respect to a park means that it is in a condition that is characteristic of its natural unimpaired state and likely to persist. Monitoring fragmentation and the rate at which it occurs in parks is therefore important to ensure that the composition of native ecosystems, rates of change and supporting processes is characteristic of the parks’ natural regions.

However, although landscape fragmentation and connectivity concepts and corresponding methods for their quantification are often used in conservation and biodiversity management, academic literature does not tend to be consistent with their definitions or what the appropriate available methods to measure these are (Calabrese & Fagan, 2004; Rayfield et al., 2011). The literature review of this thesis systematically assesses which definitions and methods are most appropriate for use in this study, followed by an overview of national parks in Canada and globally, and the importance of measuring fragmentation as an ecological indicator.

This study measures the fragmentation of Canadian National Parks at key time-steps throughout their history, including before their designation as a park or date of conservation, and up to the current day. A Progressive-Change Before-After Control-Impact Paired Series (Thiault et al., 2017) study design was used to evaluate sites by comparing the changes in the environmental conditions before and after the designation of an area as a national park against control areas that have not had the same level of protection as the national parks have. This approach focuses on how effective the designation and management of national parks have been in controlling landscape fragmentation over the 90-year time period.

This study seeks to address the following research questions (with related hypotheses that are driving this study):

1. How do fragmentation levels and rates of change compare before and after the establishment of National Parks in Canada?
2. How do fragmentation levels and rates compare between the protected areas and the nearby non-protected (control) areas?

Hypothesis: *the fragmentation levels and rates in the protected areas have been much lower than in the control areas.* As described in the literature review, common causes of landscape fragmentation in general are urbanization, roads, and related anthropogenic infrastructure, industrial development and an increase in agricultural lands. Since tourism is a key part of Canada's national parks (Bath & Enck, 2003; Benidickson, 2011), the related human development and activities are likely to have a fragmenting effect on the so-called 'natural' landscapes of federally-protected areas (CPAWS, 2019).

3. How do the lengths of time of protection of national parks relate to their levels and rates of fragmentation?

Hypothesis: *the longer a national park has been protected, it will have higher landscape connectivity than parks that have been protected for less time.*

4. How does the remoteness of parks from human activities affect their fragmentation levels?

Hypothesis: *a park is likely to be more fragmented than other regions if it has higher tourist numbers, a longer history of human influences in the parks and a greater number of transport links leading into the park.* A general prediction would be that Canada's federally-protected areas are successful in maintaining ecological integrity, thus fragmentation levels have not increased.

Quantitative information about levels and trends in fragmentation and connectivity is important for understanding their effects on current ecological processes, ecosystem services, and potential future anthropogenic development in parks and in their regional vicinity. The findings of this analysis offer insight into how human impacts have shaped the landscape in national parks in comparison to similar but unprotected areas. They provide important context for how ecological connectivity can be addressed in conservation programs across the National Park System (NPS) and in partnership with other organizations. Failures, successes and nuances can be identified and lessons about the long-term effectiveness of past and present protection strategies can be learned.

2 - Literature Review

2.1 - Section 1

Both landscape fragmentation and connectivity are fundamental concepts of landscape ecology, especially the study of wildlife movement and population dynamics. Therefore, they are relevant to conservation efforts by Canadian National Parks and Park Reserves, because Parks Canada's current responsibility is to maintain or restore the ecological integrity of its national parks, where ecosystem structures and functions are "unimpaired and likely to persist" (Parks Canada, 2019b).

2.1.1 What is landscape fragmentation?

There are a wide variety of definitions of landscape fragmentation, habitat fragmentation and fragmentation *per se*, and therefore clarity is key as to which versions are used in which contexts. An unclear definition of fragmentation could result in the mismanagement of protected areas. In most of these concepts, habitat 'patches' are referred to as areas where a habitat is divided into "useable patches which are separated from one another" by non-useable habitat, known as the "matrix" (Fahrig & Merriam, 1985; Forman, 1995). In table 1, definitions of landscape fragmentation, habitat fragmentation and related concepts are collected. There are five concepts relating to fragmentation, including two words – 'fragmentate' and 'fragment' – that are used in everyday language.

Table 2.1.1- *Definitions of "fragmentation" and corresponding subtypes from the English and German literature. (Translations: JAGJ), modified and revised from Freeman-Cole & Jaeger, in progress.*

Concept	Definition	Source
Landscape fragmentation	"Fragmentation of the landscape produces a series of remnant vegetation patches surrounded by a matrix of different vegetation and/or land use."	Saunders et al. (1991)
	"Breaking up of a habitat, ecosystem or land use type into smaller parcels" "the breaking up of large habitat or land areas into smaller parcels"	Forman (1995a: 138) Forman (1995b: 406)
	"Landscape fragmentation results from patchwork conversion and development of sites, e.g., into settlements or other intensively used areas, and from linkage of these sites via linear infrastructure. … In this paper, … 'fragmentation' shall be used as a … notion for all six phases [perforation, incision, dissection, dissipation, shrinkage, and attrition; Fig. 2]."	Jaeger (2000: 115)
Habitat fragmentation	"the process of subdividing a continuous habitat into smaller pieces, … Habitat fragmentation has three major components, namely loss of the original habitat, reduction in habitat patch size, and increasing isolation of habitat patches..."	Andrén (1994)
	"A process during which a large expanse of habitat is transformed into a number of smaller patches of smaller total area, isolated from each other by a matrix of habitats unlike the original"	Wilcove et al. (1986: 237)
	"A process whereby a contiguous patch of habitat is transformed into a number of smaller, convoluted and/or disjunct patches, isolated from each other by a matrix of habitat unlike the original"	Wang et al. (2014: 634-635)
Habitat fragmentation *per se*	"The breaking apart of habitat, independent of habitat loss"	Fahrig (2003: 487 & 509)
Landscape dissection (Landschafts-zerschnei-dung)	"Landscape dissection denotes a separation of existing (grown) ecological interconnections between spatially connected parts of the landscape. The main causes are anthropogenic, predominantly linear structures (primarily roads, railways, and power lines) that bring with them barrier effects, emissions, or collisions as well as aesthetic disturbances."	Jaeger, Grau und Haber (2005: 98)

	German original: “Landschaftszerschneidung bezeichnet ein Zertrennen von gewachsenen ökologischen Zusammenhängen zwischen räumlich verbundenen Landschaftsbereichen. Hauptursache sind vom Menschen geschaffene, vorwiegend linienhafte Strukturen (vor allem Straßen, Bahnlinien und Leitungstrassen), mit denen Barriere-, Emissions- oder Kollisionswirkungen sowie ästhetische Beeinträchtigungen verbunden sind.”	
	“Landscape dissection denotes a tearing apart of existing (grown) ecological interconnections between spatially separate parts of the landscape.” German original: “Landschaftszerschneidung bezeichnet ein Zerreißen von gewachsenen ökologischen Zusammenhängen zwischen räumlich getrennten Bereichen der Landschaft.”	Haber (1993: 62)
	“Anthropogenic dissection can be described in structural regard as predominantly linear structures or material flows created by humans that bring with them barrier effects, emissions, or collisions as well as aesthetic disturbances.” German original: “Die anthropogene Zerschneidung kann in struktureller Hinsicht als vom Menschen geschaffene, vorwiegend linienhafte Strukturen oder Materieströme beschrieben werden, mit denen Barriere-, Emissions- oder Kollisionswirkungen oder ästhetische Beeinträchtigungen verbunden sind.”	Jaeger (2003), based on Grau (1997), Walz and Schumacher (2000)
fragmentate	“to break or fall into pieces”	Webster’s New Encyclopedic Dictionary (1993)
fragment	“fragment: *noun* a small part of sth that has broken off or comes from sth larger; *verb* to break or make sth break into small pieces or parts.”	Oxford Advanced Learner’s Dictionary (2000)

Landscape fragmentation and habitat fragmentation differ in that landscape fragmentation refers to the breaking apart of a “habitat, ecosystem or land use type” (Forman, 1995) into smaller pieces. Although different land cover types can be used as habitat proxies, in comparison, habitat fragmentation is the breaking apart of the habitat of a species into smaller, potentially more convoluted patches that are isolated from each other (Wang et al., 2014; Valente et al., 2023). These patches may be linked by corridors or be within a species’ movement distance, which affects the patches’ connectivity (Foreman, 1995a). The term ‘habitat fragmentation’ differs from ‘landscape fragmentation’ and is more specific because it refers to the habitat of particular species. Like landscape fragmentation, for some authors, habitat fragmentation includes the loss of habitat, while others consider ‘habitat fragmentation per se’ and habitat loss as independent concepts (Fahrig 1999; 2002; 2003; 2017; Valente et al., 2023).

The first studies of fragmentation –referring to habitat fragmentation- that were published by Curtis (1956) and Moore (1962) conceptualized habitat fragmentation as a concept caused by habitat loss. Critically, these researchers distinguished between situations in which habitat removal either increases or does not increase the number of habitat patches. For example, when an entire habitat patch is removed from a landscape, habitat fragmentation does not occur because the number of patches has decreased rather than increased. This leads to the question of whether the effect of a loss of habitat on biodiversity differs between situations in which fragmentation stays the same, decreases or increases.

Therefore, the difference between 'fragmentation *per se*' and fragmentation more generally should be noted. The term 'habitat fragmentation' is often used interchangeably to refer to both changes in the amount of habitat and changes in the spatial configuration of the landscape (Fletcher et al., 2018). Many landscape metrics respond to habitat amount, so it is necessary to know which metrics include habitat amount and which measure configuration as an independent function to habitat amount (Wang et al., 2014). Fahrig (2003; 2017) reviewed landscape-scale studies regarding the effects of fragmentation *per se* and found that the effects of habitat fragmentation *per se* are generally much weaker than the negative effects of habitat loss on biodiversity. However, other studies have shown that at finer spatial scales, habitat amount may not be as significant as habitat fragmentation on ecosystem functioning (With et al., 2021). Overall, the effects of habitat amount in comparison to fragmentation tend to be context-specific, and so it is key to know whether a metric in use is covering landscape composition, or purely configuration when making conclusions for conservation and management (Chetcuti et al., 2020).

In a general discussion, the idea of fragmentation as a *process* rather than a *state* is prevailing. This process view implies that landscape fragmentation fundamentally includes some habitat loss, since fragmentation without change in habitat amount appears counterintuitive (Chetcuti et al., 2020). Therefore, the concept of fragmentation *per se* is controversial, but can be of importance in efforts to distinguish the different effects of landscape composition and configuration on wildlife populations (Fletcher et al., 2018; Saura, 2021; Rios et al., 2021; With et al., 2021). It is important for studies to specify which definition of fragmentation is used and at what scale, for clarity for land managers and policymakers developing conservation strategies or planning protected areas (Fletcher et al., 2023; Valente et al., 2023).

The two terms landscape fragmentation (“Landschaftsfragmentierung”) and landscape dissection (“Landscaftszerschneidung”) are often used synonymously, but their emphasis differs: fragmentation emphasizes the mosaic of area changes, such as natural to commercial and residential areas, and dissection emphasizes the network of linear and areal land uses, such as the network of transportation infrastructure and urban areas (Jaeger, 2003; Jaeger, Grau & Haber, 2005).

In this study, the definition of **landscape fragmentation by Jaeger (2000: 115)** was used:

> “Landscape fragmentation results from patchwork conversion and development of sites, e.g., into settlements or other intensively used areas, and from linkage of these sites via linear infrastructure. ... In this paper, ... ‘fragmentation’ shall be used as a ... notion for all six phases [perforation, incision, dissection, dissipation, shrinkage, and attrition; Fig. 2].”

The reason for this is that all six “phases” of fragmentation will be considered in the calculation of fragmenting elements as a barrier to connectivity. Not only linear infrastructure that dissects a landscape will be considered as a fragmenting element; perforating elements and elements that shrink the available landscape will also be considered, such as buildings and agricultural land.

2.1.2 Common causes of landscape fragmentation

In recent years there has been an increase in urbanisation and related development, such as building roads, railroads, pipelines, and water alterations, in Canada and globally (EEA & FOEN, 2011). Roads and railroads are particularly important indicators of anthropogenic environmental change because natural landscapes are cleared in a linear fashion to make way, and they open up areas for further anthropogenic influence (Fenech et al., 2005; Ament et al., 2008; Gimmi et al., 2011). Fragmentation of this kind has been increasing at a rapid pace in many parts of Canada (Crist et al., 2005; Colpitts, 2012); for example, the total extent of built-up areas of Canadian census metropolitan areas increased by 157% from 1971 to 2011 (Statistics

Canada, 2016). Furthermore, there has been intensification of agriculture and its related effects across Canada, including overgrazing and forest encroachment, along with land use change related to mining and oil, gas, and coal extraction, and other industry (Roch & Jaeger, 2014; Belote et al., 2017). This demonstrates land use change away from urban areas, and exacerbates natural land conversion to new rural communities and other private lands which is particularly prevalent around protected areas (Mcdonald et al., 2008; Gimmi et al., 2011; Adhikari & Hansen, 2018). As seen in Europe, high fragmentation values tend to be found in and around urban areas and along transport corridors (Lawrence et al., 2021). In contrast, lower levels of anthropogenic fragmentation tend to be observed in mountainous or particularly remote areas (Munroe et al., 2007; Mackey et al., 2008; EEA & FOEN, 2011). However, the exact rate of fragmentation in both non-protected and protected areas in Canada is unknown (Soverel et al., 2010).

In addition to anthropogenic influences, some common natural fragmenting elements include large rivers and other water bodies, changes in geological features and high mountains (Munroe et al., 2007; Girvetz et al., 2008). Depending on the species, these features can present a significant barrier to animal movement and overall landscape connectivity (Stronen et al., 2012; Dutcher et al., 2020).

2.1.3 What is landscape connectivity?

Depending on the definition used, the metrics applied by researchers to quantify connectivity will differ. Thus, "the same landscape may have different landscape connectivity values when different measures of landscape connectivity are used" (Goodwin & Fahrig, 2002; 552). Therefore, when quantifying the connectivity of a region or analysing a previous study, it is important to recognize which concept of connectivity is being used and in which context.

Table 2.1.3- *Review of definitions of "connectivity" and corresponding subtypes from the literature (modified and extended after Meiklejohn et al., 2009).*

Concept	Definition	Source
Connectivity	Connectivity "measures the processes by which the sub-populations of a landscape are interconnected into a demographic functional unit."	Merriam (1984: 6)
	"The extent to which patches are connected to one another by similar habitat or corridors"	Noss (1987: 160)
	"The integration of populations into a single demographic unit."	Horskins et al. (2006: 641)
Landscape connectivity	"Landscape connectivity is the degree to which the landscape facilitates or impedes movement among resource patches. … Landscape connectivity can be measured for a given organism using the probability of movement between all points or resource patches in a landscape."	Taylor et al. (1993: 571-572)
	"A species-specific characteristic determined by the interaction between the movement potential of each species and landscape structure. "	Mönkkönen & Reunanen (1999: 302)
	Landscape connectivity "encapsulates the combined effects of (1) landscape structure and (2) the species' use, ability to move and risk of mortality in the various landscape elements, on the movement rate among habitat patches in the landscape."	Tischendorf & Fahrig (2000: 8)
	"Consists of both the response of individuals to landscape features (behaviour) and the patterns of gene flow that result from those individual responses."	Brooks (2003: 433)
	"The extent to which a species or population can move among landscape elements in a mosaic of habitat types."	Hilty et al. (2006: 90)
	"Landscape connectivity thus combines a description of the physical structure of the landscape with an organism's response to that structure."	Taylor et al. (2006: 29)
	Landscape connectivity describes "the ease with which these individuals can move about within the landscape" as a function of the organism's behavioural response to landscape elements and the spatial configuration of the entire landscape.	Kindlmann & Burel (2008: 879)
	"An anthropogenic construct, and refers broadly to the connectedness of vegetation cover on the landscape."	Pelletier et al. (2017: 2)
Habitat connectivity	"The functional relationship among habitat patches, owing to the spatial contagion of habitat and the movement responses of organisms to landscape structure."	With et al. (1997: 151)
	"The degree of functional connectivity between patches of optimal habitat for individual species"	Correa Aryam et al. (2016: 8)
	"A species-specific concept defined as the potential for movements between habitat patches, and quantified at either patch or landscape scales"	Pelletier et al. (2017: 2)
Structural connectivity	"The spatial contagion of habitat."	Mönkkönen & Reunanen (1999: 302)
	"Derived from physical attributes of the landscape, such as size, shape and location of habitat patches, but does not factor in dispersal ability."	Calabrese & Fagan (2004: 530)
	"Describes the shape, size and location of features in the landscape."	Brooks (2003: 433)
	"Structural connectivity ignores the behavioural response of organisms to landscape structure and describes only physical relationships among habitat patches such as habitat corridors or inter-patch distances."	Taylor et al. (2006: 30)
	"Describes the physical relationships among habitat patches while ignoring the behavioural response of organisms to landscape structure".	Kadoya (2009: 6)
	"A product of amount of habitat, spatial configuration and condition across multiple scales."	Andersson & Bodin (2009: 123)

	"*Structural connectivity* refers to the physical relationship between landscape elements."	Meiklejohn et al. (2010)
	"[incorporating] spatial position"	Kool et al. (2013: 167)
	"Spatial relationships (continuity and adjacency) between the structural elements of the landscape… which is independent on the ecological characteristics of the species"	Correa Aryam et al. (2016: 8)
Functional connectivity	"[incorporating] spatial position, dispersal, behaviour, habitat quality"	Kool et al. (2013: 167)
	"Functional connectivity, on the other hand, increases when some change in the landscape structure (including but not limited to changes in structural connectivity) increases the degree of movement or flow of organisms through the landscape. The original concept of landscape connectivity thus emphasizes the functional connectivity of landscapes."	Taylor et al. (2006: 30)
	"*Functional connectivity* describes the degree to which landscapes actually facilitate or impede the movement of organisms and processes. Functional connectivity is a product of both landscape structure and the response of organisms and processes to this structure. Thus, functional connectivity is both species- and landscape-specific."	Meiklejohn et al. (2010)
	"The links are not interpreted as structural features of the landscape or as corridors, but rather as the organism might experience them."	Galpern et al. (2011: 45)
	"Landscape features that facilitate or impede the movement of species between habitat patches"	Correa Aryam et al. (2016: 8)
Actual connectivity	"Relates to the observation of individuals moving into or out of focal patches, or through a landscape, and thus provides a concrete estimate of the linkages between landscape elements or habitat patches."	Calabrese & Fagan (2004: 530)
Potential connectivity	"Combines ... physical attributes of the landscape with limited information about dispersal ability to predict how connected a given landscape or patch will be for a species"	Calabrese & Fagan (2004: 530)
Genetic connectivity	"The hard-to-observe process of movement of individuals through a landscape can be inferred by examining genetic structure and relatedness"	Dutcher et al. (2020: 289)
	Also known as biological connectivity - "the actual movement of individuals and their genes between populations in the landscape"	Brooks (2003: 435)
	"[the premise of landscape connectivity] can also be applied in different processes like gene flow"	Correa Aryam et al. (2016: 8)
Ecological connectivity	"The connectedness of ecological processes, such as energy flow through an interaction network wherein species are connected via trophic relationships"	Pelletier et al. (2017: 2)
Population connectivity	"[the premise of landscape connectivity] can also be applied in different processes like … dispersal across discrete populations"	Correa Aryam et al. (2016: 8)
Between-patch connectivity / Inter-patch connectivity	"Non-contiguous habitat patches may functionally be connected if the species can cross the non-habitat area (matrix) successfully and move between habitat patches."	Tischendorf & Fahrig (2000: 8 & 10)
	"the area made available by the connections among different habitat patches (i.e. interpatch connectivity)"	Saura & Rubio (2010: 524)
	"between-patch connectivity, i.e. the connectivity between different habitat patches, … Habitats are commonly delineated as discrete patches and connectivity is often based on the emigration and immigration between them. This concept can be described as *between-patch connectivity*, or *inter-patch connectivity*, since the movement within the patches is not considered."	Spanowicz & Jaeger (2019: 2261-2262)

Within-patch connectivity / Intra-patch connectivity	"When the organism's movement is confined to its preferred habitat, i.e. individuals so not cross the habitat/matrix boundary, and the organism moves freely within the preferred habitat"	Tischendorf & Fahrig (2000: 8)
	"the connected habitat area that exists within the patches (i.e. intrapatch connectivity)"	Saura & Rubio (2010: 524)
	"connectivity within habitat patches"	Spanowicz & Jaeger (2019: 2262)

Although the most often cited definition of landscape connectivity is that of Taylor et al. (1993), it is not the earliest. The first clear definition of "connectivity" in a landscape context was proposed by Merriam (1984), and roughly describes what would later be considered as "functional connectivity" due to its information on how "sub-populations", i.e., animals or plants, are connected in a "functional unit". The definition of connectivity by Horskins et al. (2006) is similar, but less precise. Overall, definitions for landscape connectivity tend to include a description of the landscape's structure, as well as an individual's ability to move through said landscape, including Taylor et al. (1993), Tischendorf & Fahrig (2000), and Taylor et al. (2006). However, Kindlmann & Burel (2008) only note that landscape connectivity is the ease at which an individual moves within a landscape.

An interesting take on the definition of landscape connectivity is provided by Pelletier et al. 2017), who describe it broadly as the "connectedness of vegetation cover on the landscape". This implies that landscape connectivity refers to vegetation cover, whereas habitat connectivity refers to "the potential for movements between habitat patches", whether that is animal movement or gene flow, for example.

The concept of habitat connectivity is generally used in a very similar sense to that of landscape connectivity as defined by Taylor et al. (1993), Tischendorf & Fahrig (2000) and Taylor et al. (2006), in that it describes animal movement across a landscape. Like habitat and landscape fragmentation, the main difference between habitat rather than landscape connectivity is that it is species-specific and distinctly refers to the ability of an organism of a certain species to move between patches in a landscape where it can successfully live. The same landscape will usually provide distinct levels of habitat connectivity to different species because (a) their respective habitat type differs and (b) the resistance of movement to the other landscape elements will differ between species.

The term “movement”, in relation to animals, refers to several types of movement. At least three types of wildlife movement should be considered when assessing definitions of connectivity: daily movement within an individual’s home-range (e.g., for foraging), seasonal migrations (round-trip relocation), and dispersal (Johnson et al., 2002; Jacobson and Peres-Neto, 2010). In addition, range shift in response to climate change-induced habitat distribution and gene flow can also be considered as “movement”. An interesting observation is that Taylor et al. (2006) write that the “original concept of landscape connectivity emphasizes the functional connectivity of landscapes”, referring to both the physical structure and species movement behaviour. All the definitions of landscape connectivity correspond with this observation, other than that of Pelletier et al. (2017).

The two most used ‘sub-concepts’ of connectivity in a landscape are structural connectivity and functional connectivity. In general, structural connectivity refers to the physical aspects of landscape structure and does not consider the movement response of animals (Calabrese & Fagan, 2004; Taylor et al., 2006; Correa Aryam et al., 2016). It is usually assumed that animals can move through structurally connected, or contiguous, areas of habitat such as corridors and patches, no matter what the land cover in these habitats. In contrast, functional connectivity does consider the movement response of the organisms of interest. It has been argued that there is no requirement for a habitat to be structurally connected in order for it to be functionally connected. For example, some species are capable of moving across a matrix up to a certain distance due to their gap-crossing abilities (Taylor et al., 2006; Chetcuti et al., 2021b).

A similar but not equal concept to functional connectivity is *actual connectivity*, which incorporates animal movement that has already been observed and measured. Measuring this type of connectivity generally requires some form of *track and trace* data collection, which can both be extremely costly and time-consuming (Calabrese & Fagan, 2004; Poli et al., 2020). Alternatively, *genetic connectivity* can be measured a few generations after movement as part of metapopulation studies to demonstrate how animal populations and their genes have moved across a landscape in comparison to genetically similar populations nearby (Dutcher et al., 2020). However, the genetic effects of habitat loss or fragmenting barriers are usually only observable after a significant lag time of many generations. In contrast, *potential connectivity* incorporates animal movement by the predicted effect the landscape will have on a species’ movement, depending on a species' dispersal distance and potentially other factors, which is often more time- and cost-effective for landscape-scale studies.

For *connectivity,* it is even less clear than for *fragmentation* whether the term refers to landscape configuration or to a combination of both landscape configuration and composition. A term such as "habitat connectivity *per se*", which would indicate landscape configuration, has not been used in the literature. The definitions in Table 2.1.3 seem to refer to both landscape configuration and composition.

Two other key concepts in the field are *between-patch* and *within-patch connectivity*. To gain an accurate measurement of landscape connectivity, *within-patch* must be considered, however many metrics do not acknowledge it. A number of metrics only measure *between-patch connectivity*, which can produce misleading results such as scenarios where connectivity increases with greater habitat loss and fragmentation (Spanowicz & Jaeger, 2019). Applying misleading results from metrics that do not consider within-patch connectivity can have detrimental effects on natural ecosystems, because a reduction in within-patch connectivity by human activities such as fragmentation is neglected.

This comparison of definitions demonstrates that "connectivity" is no single concept but refers to a wide range of concepts that differ from each other in several important ways. This study used the **definition of functional landscape connectivity from Taylor et al. (2006:30**) in its analysis, because fragmenting elements are considered as barriers with different strengths impeding animal movement:

> "Functional connectivity, on the other hand, increases when some change in the landscape structure (including but not limited to changes in structural connectivity) increases the degree of movement or flow of organisms through the landscape. The original concept of landscape connectivity thus emphasizes the functional connectivity of landscapes."

In addition, both between-patch and within-patch connectivity were considered. The metric used to measure connectivity and fragmentation in this study was the Effective Mesh Size (m_{eff}), a spatial pattern metric (Jaeger, 2000, 2003; Calabrese & Fagan, 2004).

2.1.4 What is the link between fragmentation and connectivity?

At first glance, the concept of increased fragmentation seems to intrinsically imply a reduction in connectivity in the remaining ecological network, and in both the literature and in conservation practice, connectivity and fragmentation are often viewed as inversely related concepts. It is commonly thought that maintaining or restoring the connectivity between habitat patches will mitigate landscape/habitat fragmentation, and preserve or restore associated ecological processes, including animal movement (Jordán, 2003; Galpern et al., 2011). Although this outlook may be correct to some extent, it often fails to take into account the concept of within-patch connectivity and the animals' ability to move across the matrix between habitat fragments (Rayfield et al., 2011; Spanowicz & Jaeger, 2019; Chetcuti et al., 2021b).

The relationship between the two concepts is that fragmentation *per se* is equivalent to a reduction of within-patch connectivity and vice-versa: (a) Fragmentation *per se* always results in the breaking of within-patch movement connections that existed within the patches, that are now separated into pieces; and (b) provided there is no habitat loss, any decrease in within-patch connectivity implies there is an increase in fragmentation *per se,* because that is the only way that this can happen.

In contrast, between-patch connectivity may or may not increase when patches are broken up (no habitat loss), depending on the distance between the resulting fragments. Thus, distances between the fragments have an important effect on between-patch connectivity, but they do not matter for neither within-patch connectivity nor fragmentation *per se.* This is summarised in figures 2.1.4a and b:

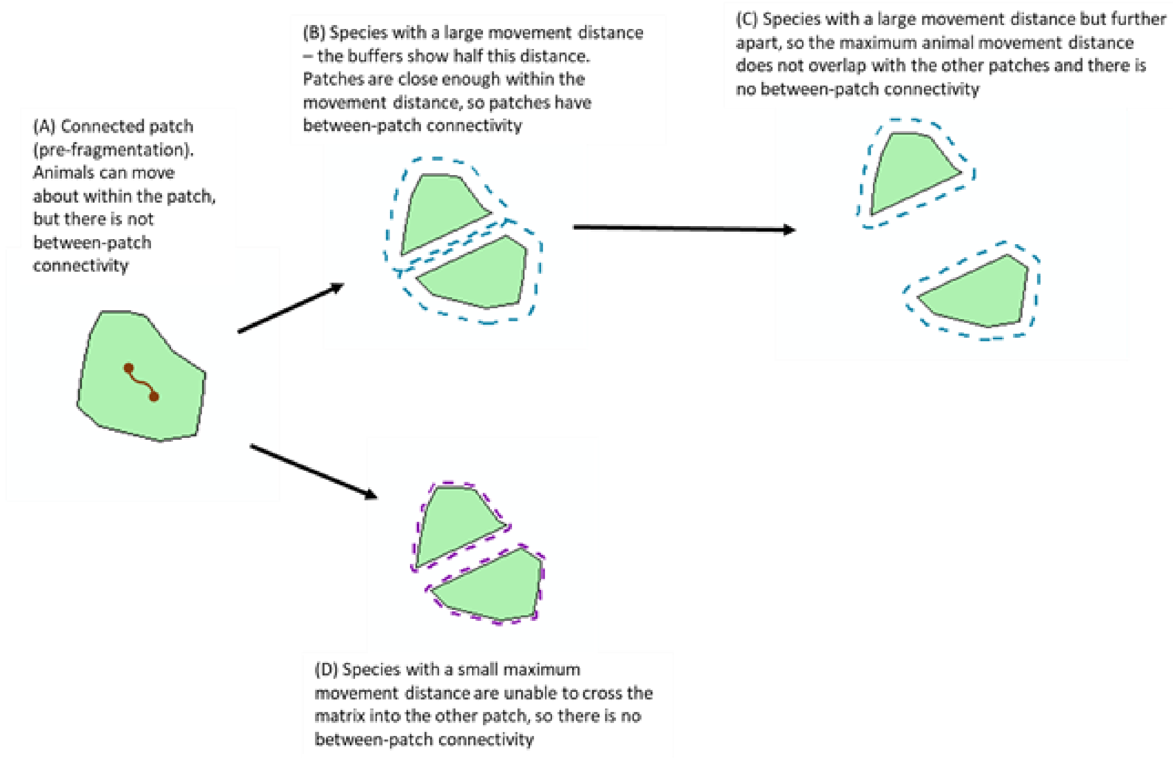


Figure 2.1.4a - *Changes in between-patch connectivity as a consequence of fragmentation per se and increasing distance between the fragments. The buffers indicate half of the maximum movement distance of the target species. The brown line indicates an example of a connection between two points located within habitat (within-patch connectivity in (A) and between-patch connectivity in (B)).*

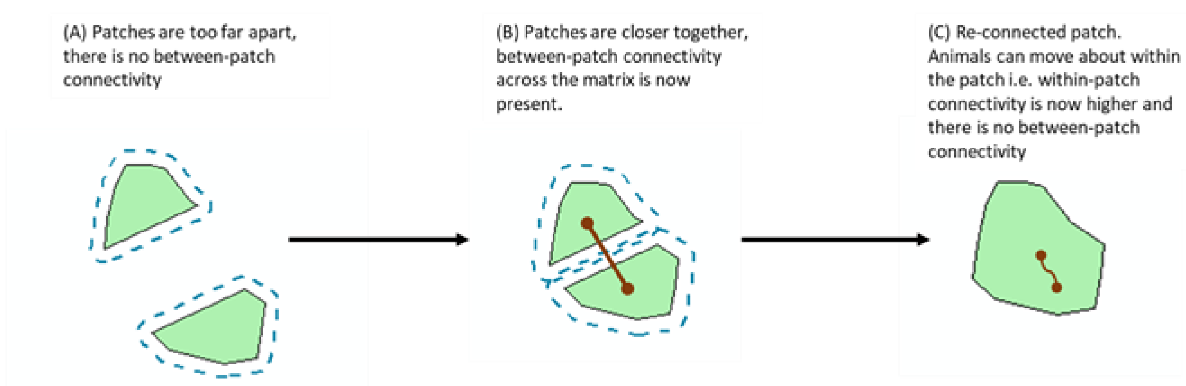


Figure 2.1.4b - *Increasing between-patch connectivity and within-patch connectivity as a consequence of decreasing distance between the fragments (B) and connecting the fragments (C).*

However, it is not just a change in habitat configuration that can cause an increasing distance between patches. Habitat loss changes landscape composition and can also cause a break-up of habitat patches, for example the construction of a road or an urban area, creating a barrier that impedes animal movement. This process is of key investigation in this study. Habitat loss often increases habitat-inter-patch distances and decreases patch sizes (Goodwin & Fahrig, 2002). Therefore, the decrease in habitat that dissects or subdivides a patch, changes both within-patch and between-patch connectivity, depending on the strength of the barrier against animal movement.

2.1.5 What are the effects of landscape fragmentation on biodiversity in general?

Research on the effects of habitat fragmentation on biodiversity is very diverse, especially considering the wide variety of methods employed to measure landscape fragmentation and connectivity (Fahrig, 2003; Kool et al., 2013; Fletcher et al., 2016). Landscape fragmentation in its wide sense (including habitat loss) is a major factor causing biodiversity loss, particularly due to the increase in anthropogenisation of landscapes. The disruption caused by fragmentation has been described as a "threat to landscape quality and to the sustainability of human land-use" (EEA & FOEN, 2011: 52), including ecosystem services and perceived landscape value (Wartmann et al., 2021). Remaining unfragmented area of the landscape (the "landscape meshes") are becoming even smaller (Jaeger et al., 2007), which negatively affects wildlife populations by reducing habitat extent and quality, reducing species richness, and subdividing populations into "smaller and more vulnerable fractions" and reducing genetic variability (Jaeger et al., 2008: 738; Roch & Jaeger, 2014; Deslauriers et al., 2018).

Impacts on the environment include a change in land cover, the subsequent changes in local climate, emissions from roads and railroads, the modification of watercourses and disruption to various other ecological processes. In addition, linear fragmenting features such as roads and railways facilitate the spread of invasive non-native species, which disrupts natural ecosystem processes and increases native species populations' vulnerability (Jaeger et al., 2007). As well as roads acting as barriers to movement and reducing the amount of habitat and associated connectivity, there is also an enhanced species mortality risk due to vehicle collisions, which further adds to a species' vulnerability to extinction (Fahrig & Rytwinski, 2009; Proctor et al., 2012; Stronen et al., 2012). The concept of ecological integrity (as described in section 2) in Canada is thus threatened due to fragmentation of landscapes (Canadian National Parks Act, 2000).

In addition, climate change exacerbates the effects of urbanisation and intensification of agriculture, and thus the effects of landscape fragmentation. Not only does this disrupt natural ecosystem processes, but it also increases the impact of phenomena such as urban heat islands (Tan et al., 2010) and drought/desertification (UNDP, 2020). It also encourages the dispersal of species due to a relocation or removal of appropriate habitat (Moser et al., 2007; Jacobson and Peres-Neto, 2010). Many species have long response times to changes in landscape structure (known as lag time or extinction debt), which may also make it difficult to analyse how the recent boom in anthropogenic activities in Canada have affected certain species (EEA & FOEN, 2011; Dutcher et al., 2020).

2.2 - Section 2

2.2.1 An overview of national parks in Canada

"The national parks of Canada are hereby dedicated to the people of Canada for their benefit, education and enjoyment, subject to this act and the regulations, and the parks shall be maintained and made use of so as to leave them unimpaired for the enjoyment of future generations" Canada National Parks Act, 2000, Section 4(1).

Human perspectives towards national park areas have been as varied throughout history as they are today. Currently, the overarching motive behind Canadian national parks is to "protect natural environments representative of Canada's natural heritage" (Parks Canada Agency, 2021d). Canada has a total of 48 terrestrial national parks and national park reserves, making up the country's National Parks System (Figure 2) and covering more than 340,000km^2 of its land (Mcnamee et al., 2019). These federally protected areas are governed by Parks Canada and are created from and managed through guidance and policy statements (Benidickson, 2011). As shown in Figure 2, there are 39 terrestrial "Natural Regions" in the country, with the aim of having a representative federally protected area in each one, whether this be a National Park or a Park Reserve (Warner, 2008; Benidickson, 2011). In general, a representative National Park will portray the native “geology, physiography, vegetation, wildlife and ecosystem diversity” of the region, and either be in a healthy, natural state or have the potential to be restored to a natural state (Parks Canada, 2008:1.1.1i). Currently, there are 9 natural regions not yet represented in the National Parks System (NPS).

A Park Reserve distinct from a national park in that it is an area or a portion of an area proposed to become a national park which is subject to an unresolved claim of Aboriginal rights which will be negotiated by the government (Canada National Parks Act, 2000, s.2(1)). It is otherwise mostly treated the same as a national park, and local Indigenous communities may continue with traditional hunting, fishing, trapping and spiritual activities (Dearden & Berg, 1993; Murray, 2010).

In conjunction with the goal of leaving national parks “unimpaired for future generations”, the concept of maintaining ‘ecological integrity’ as a management standard has emerged since the Parks Policy of 1994 (Warner, 2008; Benidickson, 2011). Parks Canada states that an ecosystem has ecological integrity when a) “it has the living and non-living pieces expected in its natural region” and b) “its [ecosystem] processes occur with the frequency and intensity expected in its natural region.” (2019). Therefore, the agency has the main objective to allow people to enjoy national parks without damaging their ecological integrity, by maintaining or restoring the expected genes, species, and communities of a region, striking a delicate balance between tourism and conservation. Since Canada is a member of the International Union for Conservation of Nature (IUCN), Parks Canada contributes towards the development of internationally accepted standards for parks worldwide, and is responsible for the country’s UNESCO obligations. In addition, some of Canada’s National Parks have been designated World Heritage Sites, and others make up portions of biosphere reserves as part of the UNESCO Man and Biosphere Programme (McNamee, 2019).

However, the Canadian NPS has not always been managed with the goal of ecological integrity in mind. The original national parks were created purely as tourist areas, and since then management and designation standards have shifted every few decades (Dearden & Berg, 1993; Wiersma & Nudds, 2009).

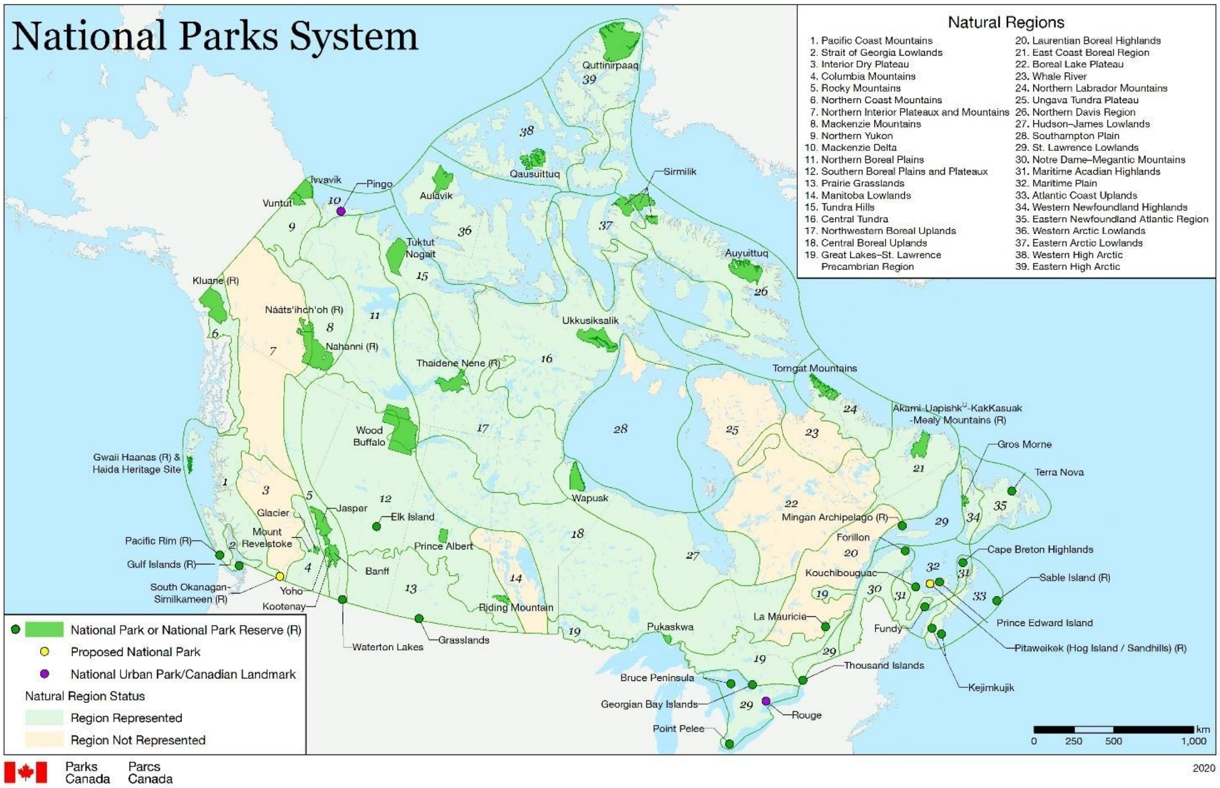


Figure 2.2.1 *- National Parks System map including the 39 Natural Regions of Canada and existing and proposed new parks (Parks Canada, 2021d)*

2.2.2 Globally, how effective have protected areas been in curbing fragmentation?

"Nothing dollarable is safe, however guarded." - John Muir, 1908

The creation of protected areas (PAs) at a certain location and time are a result of interactions between environmental factors, land-use history, and socio-economic and political context (Gimmi et al., 2011). PAs are often viewed as the cornerstones of biodiversity conservation; however, a variety of reasons ranging from differing management strategies to financial ability to enforce protection, can render certain PAs ineffective in preventing increasing human pressures within and on their boundaries (Mcdonald et al., 2008; Gimmi et al., 2011; Lawrence et al., 2021; Kubacka et al., 2022). It has been noted that, globally, a major common driver of species presence in an area, and thus species sensitivity to fragmentation, is the size of habitat patches (Keinath et al., 2017). Anthropogenic land use pressures on PAs vary from country to country and can include: encroaching urbanization and suburbanization, expanding transportation systems, intensive agriculture, deforestation, and tourism infrastructure. The majority of these pressures lead to the landscape fragmentation of protected areas in some form. In addition, it has been suggested that infrastructure development and the resulting fragmentation is a good indicator for park effectiveness in comparison to an indicator such as deforestation, which is less likely in the global north (Gimmi et al., 2011).

Some anthropogenic pressures on PAs occur outside of their boundaries but create significant changes to the 'protected' ecosystems within (Dearden & Berg, 1993; Leroux & Kerr, 2013; Adhikari & Hansen, 2018). The effectiveness of PAs often depends on how well connected they are to viable habitats adjacent to their boundaries (Custode et al., 2023). A key model that describes this process is the Boundary Model (Schonewald-Cox & Bayless, 1986). This frames a PA's administrative boundary as a 'filter' that is 'activated' by whatever regulations on human influence are mandated and enforced. Should there be strong external pressures and/or internal pressures on a PA, a generated edge of actual ecological protection will move inward towards the centre of the PA, causing its effective size to become smaller. In contrast, effective enforcement of ecological protection mandates within a PA's administrative boundaries will enable a PA to be effective in biodiversity conservation across its entire area. The model also emphasizes how an area's shape and size will influence the effect of external pressures. The Boundary Model is important to consider when assessing the effectiveness of protected areas worldwide due its integration of human activities around PAs, attitudes to and enforcement of protection, and location of natural and administrative boundaries (Schonewald-Cox & Bayless, 1986; Dearden & Berg, 1993).

As described in the Boundary Model, the effectiveness of a PA in curbing landscape fragmentation often depends on its location and associated attitudes and ability of that place towards enforcing environmental protection from human influences. For example, there are more drivers of land clearing in PAs in Asia, Africa and South America than in Europe and North America, and therefore park management in these areas face increased difficulties (Nagendra, 2008).

Like Canada, in the United States, National Park Service management faces the challenge to balance ever-increasing visitor use with associated environmental consequences, along with unprecedented intensification of land use change outside park boundaries (Ament et al., 2008; Adhikari & Hansen, 2018). In numerous studies, it has been shown that PAs in the US have been relatively effective in preventing fragmentation so far, however increasing anthropogenic pressures are on track to cause landscape fragmentation in the future, in addition to the PAs becoming more isolated from other natural areas (Ament et al., 2008; Gimmi et al., 2011; Adhikari & Hansen, 2018). In a similar vein, national parks in Thailand need to balance tourism with ecological protection, leading to moderate fragmentation in their boundaries (Sims, 2014). However, wildlife sanctuaries in Thailand are more restrictive to human influences and are specifically aimed at preserving biodiversity, and have been shown to be more effective than the national parks in preventing fragmentation and increasing forest cover (Sims 2014). In Mexico, a similar trend has been described, with reserves that receive more financial support for their management being more successful in preventing land use change and fragmentation (Figueroa & Sánchez-Cordero, 2008). This highlights the effects of different park management and protection enforcement and how environmentally focused management styles can be successful in reducing landscape fragmentation in protected areas.

European PAs also face a loss of habitat outside of their boundaries, increasing their isolation from other natural landscapes (Santiago-Ramos & Feria-Toribio; 2021). In addition, in the main network of PAs in Europe - Natura 2000 – several studies have shown that many of these PAs are very fragmented, especially in Western Europe (Lawrence et al., 2021; Santiago-Ramos & Feria-Toribio, 2021; Kubacka et al., 2022). Like in the US, urban sprawl and infrastructure development are major causes of the fragmentation of PAs, along with unsustainable farming and forestry (Rodríguez-Rodríguez et al., 2019; Kubacka et al., 2022). Despite many PAs in the Natura 2000 receiving significant financial support, the centuries-old heavy human footprint in Europe has caused many PAs to have high landscape fragmentation (Lawrence et al., 2021). The PAs in Europe that have been designated for the longest time appear to have the most beneficial effects on natural land cover surrounding them (Mingarro & Lobo, 2023).

PAs in other areas of the world are also in danger of increasing human pressures and high fragmentation values, but for different reasons. For example, in Xishuangbanna, China, high rates of deforestation to make way for rubber plantations have led to PAs being used as a source of land for this development, despite its illegality (Sarathchandra et al., 2018). Similarly, protected areas in Togo are under pressure from illegal wood extraction and agricultural production (Diwediga et al., 2017). Like described in the Boundary Model, weak management of protected areas, for political, economic or social reasons, opens the administrative boundary ‘filter’ to pressures that lead to fragmentation (Schonewald-Cox & Bayless, 1986; Munroe et al., 2007).

A final consideration about the effectiveness of protected areas around the world in preventing landscape fragmentation is the ‘worthless lands’ hypothesis, as described by Runte (1979). This states that many PAs are established in marginal lands that are unlikely to be able to be exploited for commercial purposes. As such, PAs in remote and commercially non-valuable areas are less likely to be affected by anthropogenic landscape fragmentation, thus boosting the total area of protection in certain countries even if these locations are not a priority for environmental protection (Lawrence et al., 2021; Santiago-Ramos & Feria-Toribio; 2021). Brennan et al. (2022) show that the majority of critical connectivity areas remain unprotected, and so either new protected areas covering key movement corridors or a reduction in human impacts on these areas are necessary. However, in recent years some studies have shown that land use change and fragmentation has been occurring even in geographically remote areas such as Western Honduras and Australia (Munroe et al., 2007; Mackey et al., 2008). In addition, changing climate may push human infrastructure further into marginal lands as landscapes become more profitable and accessible (Mackey et al., 2008; Lawrence et al., 2021).

Overall, it appears that the most effective protected areas at preventing landscape fragmentation across the globe are those with the financial and political support to prevent the development of human pressures, especially in areas where anthropogenic activity is likely. Protected areas as an effective environmental management option are effective, if managed in an environmentally focused manner.

2.2.3 What are common ecological stressors for national parks in Canada?

"DO NOT LET MOOSE LICK YOUR CAR" - Jasper National Park, 2020

Despite national parks often being thought of as pristine areas of wilderness protected by their boundaries, stressors on park ecosystems can originate from both inside and outside the park boundaries (Rivard et al., 2000; Parks Canada, 2019a). They include events, actions, factors, or long-term change to natural processes that prevent ecosystems from maintaining or recovering their integrity, at a variety of timescales and levels of biological organization (Borics et al., 2013). Many stressors behave in a synergistic manner, thus having cumulative effects on the natural processes in an ecosystem (Mortimer-Sandilands, 2009; Dietz et al., 2021). Although protected areas in Canada have been found to do fairly well against habitat loss and fragmentation compared to other protected areas in other locations around the globe, they do still experience anthropogenic landscape change (Gimmi et al., 2011; Heino et al., 2015; Olsoy et al., 2016; Schulze et al., 2018).

The major stressors on national parks, as identified by Parks Canada, are:

- Habitat loss and degradation
- Habitat fragmentation/the reduction of landscape connectivity (such as the building of roads and trails)
- Losses of large carnivores and keystone species, such as wolves and bison
- Air pollution
- Pesticides
- Invasive non-native species
- Over-use of national parks by humans
- Climate change impacts and its related ecological changes (ECCC, 2019; ECCC, 2020).

In addition, parks can be affected by past and current land management processes, including fire management, wildlife translocation and dams, and even the most remote parks with lower direct human footprints can be influenced by pollution and climate change (Dietz et al., 2021). Effects on national parks that are caused by pollution and climate change include temperature change, a change in precipitation and snow patterns, acidification of water systems and permafrost thaw, which can lead to an exacerbation of the previously noted stressors on national parks due to their cumulative impacts (Lemiuex et al., 2011; Andrew et al., 2014; Dietz et al., 2021).

Different ecosystems respond in a variety of ways to stressors and management actions; some responses of the ecosystems could take many years (Zorn, 2005; ECCC, 2019). In a large country like Canada, the characteristics of national parks, their likely response to major stressors and associated management practices are strongly related to their latitude, and occasionally longitude (Rivard et al., 2000). In addition, land use in Canadian parks is often positively correlated with land use in surrounding regions (Rivard et al., 2000; Andrew et al., 2014).

Habitat fragmentation as an ecological stressor can be linked to increasing the effects of almost all the other major stressors identified by Parks Canada. Firstly, as per the landscape fragmentation vs. fragmentation per se debate (Fahrig, 2003; 2017; Fletcher et al., 2018), when fragmentation occurs in a real-world context, habitat loss also occurs (Moser et al., 2007). In addition, habitat loss increases the effects of fragmentation over time by increasing habitat patch isolation, thus demonstrating the link between the two concepts (Zeller et al., 2020). As described in the Boundary Model, isolated ecosystems and patches, or protected areas alone do not always provide the functions or materials required to sustain species movement, ecosystem processes, or genetic and species diversity, since these processes often occur on regional scales (Schonewald-Cox & Bayless, 1986; Woodley, 2010; Belote et al., 2016).

Parks Canada also identified “losses of large carnivores” as a significant ecological stressor on national parks. Landscape fragmentation in national parks can be one of the key drivers of these losses, and of losses of other species’ metapopulations (Girvetz et al., 2008; Rayfield et al., 2011). Large habitat patches are important for many species to sustain viable populations (Moser et al., 2007), particularly flagship species such as grizzly bears and wolves (DeFries 2007; Stronen et al., 2012; Parks Canada, 2019a). Landscape fragmentation reduces connectivity by creating barriers to animal movement to other patches relevant to their lifecycle, and to gene flow, as well as creating overly small habitat patches, thus greatly contributing to biodiversity loss (Moser et al., 2007; Girvetz et al., 2008).

In addition, development of linear fragmenting transportation routes and other infrastructure increases the dispersion of air pollutants, noise, and invasive non-native species (Fenech et al., 2005; Jaeger et al., 2010; Spernbauer et al., 2023). Both invasive plants and invasive animals cause problems for parks across Canada by modifying the natural ecosystem and altering park flora and fauna (Rivard et al., 2000; Crist et al., 2005; Parks Canada, 2019a). Another critical impact of fragmenting transportation elements is the direct mortality of animals hit by moving vehicles (Coffin, 2007; Fahrig & Rytwinski, 2009; Stronen et al., 2012). Transportation infrastructure can cause a form of ecological trap, since often it attracts wildlife for feeding or mineral lick reasons, as demonstrated by the risk of collisions with trains for grizzly bears in Banff National Park, (Proctor et al., 2012; St. Clair et al., 2019), and risk of collisions between moose and cars due to artificially created mineral licks from road salt run off (Rea et al., 2013).

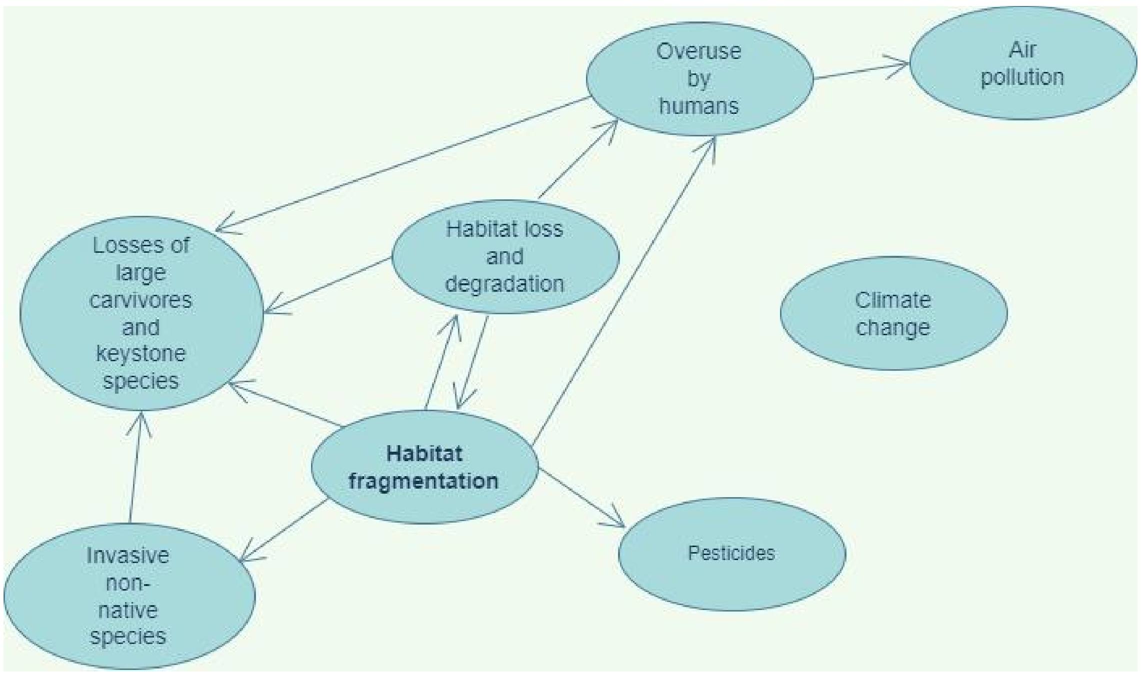


Figure 2.2.3 *- Ecological stressors in National Parks and their interactions with each other*

Underlying all the above, the main risk to maintaining ecological integrity in parks is the over-use of national parks and their surrounding areas by humans, and the landscape fragmentation that occurs with this (DeFries et al., 2007; Warner, 2008). It has been noted by non-governmental organisations that there is an increased focus on tourism, non-nature focused recreational activities, and revenue generation in the Canadian National Parks System, due to a lack of attention to the effect these might have on parks' ecological integrity and increasingly scarce governmental funding for environmental protection (Dearden & Berg, 1993; McNamee, 2010; CPAWS, 2016). Wildlife may avoid movement near roads, hiking/biking trails and towns to avoid encountering people or will have a physical barrier intended for human use in national parks, preventing them from moving between habitat patches (Moser et al., 2007; Proctor et al., 2012; Whittington et al., 2019).

2.2.4 Ecological indicators in national parks and the importance of measuring fragmentation as an indicator.

Parks Canada assesses ecological integrity of national parks by monitoring "representative" components of major park ecosystems, such as forests, freshwater, and wetlands (ECCC, 2019), through a set of ecological indicators. For each "main" ecosystem, a scientifically sound set of environmental measures is chosen, based on appropriateness, representativeness, monitoring needs, and cost-effectiveness. The selected ecosystems tend to form most of the park area and are important to the park's biological functioning (ECCC, 2020). The ecological indicators used by Parks Canada include: species richness, population dynamics of indicator species, an ecosystem's trophic structure, disturbance frequencies and size, nutrient retention and pollutants, human land-use patterns, climate and finally, habitat fragmentation through the measurement of patch size, inter-patch distances and forest interior (Welch, 2002; Parks Canada & CPC, 2008).

Using landscape and/or habitat fragmentation as ecological indicators can be considered a "level-2 indicator" (Martin & Proulx, 2020). This designation recognizes that an ecosystem does not need to be pristine or completely untouched by human activities to still be a major contributor to maintaining regional biodiversity and ecosystem functioning. Landscape fragmentation as an ecological indicator allows comparison between ecosystems in different naturalness and ecological contexts, and as such is suitable to analyze a protected area such as a park and a more developed area like a town (Jaeger et al., 2008; Martin & Proulx, 2020).

Currently, it is unknown how many parks are using landscape fragmentation as part of ecological integrity monitoring at suitable time-steps to ascertain the rate of landscape change (OAGC, 2013). Given the underlying effects landscape fragmentation has on other ecological stressors on national parks and their greater ecosystems, and its usability as an indicator to compare areas of different environments and sizes, it is important for a temporal analysis such as that in this study to be made (Martin & Proulx, 2020).

Considering that this is a time of rapid global change caused by severe anthropogenic pressures, one of the urgent key recommendations to preserve global biodiversity is to ensure representative habitat conservation, to create large conservation areas, to restore and maintain connectivity and movement corridors between these areas, and to reduce the effects of other related ecological stressors, again described in the Boundary Model (Schonewald-Cox & Bayless, 1986; Aycrigg et al., 2016; Damschen et al., 2019). The landscape matrix surrounding National Parks plays an important role in connectivity because organisms will move across them no matter their protected status (Proctor et al., 2012; Stronen et al., 2012). One of the recommended improvements in landscapes for national parks in Canada is to identify elements that favor ecosystem connectivity in parks and their greater park ecosystems (Parks Canada & CPC, 2008). For example, this is a major benefit of Canada's boreal and Arctic parks, since they provide large areas that are unaffected by human pressures, which are necessary in maintaining natural ecosystems (Andrew et al., 2014). Favoring ecosystem connectivity may also require complementary approaches alongside EI monitoring inside parks, to conserve ecosystems outside of protected areas, such as provisions in environmental assessments and future infrastructure developments (Dietz et al., 2021). By improving landscape permeability – increasing the ability of organisms to move across a landscape – between parks and other landscapes, functional connectivity will be improved (Lemieux et al., 2021).

Wildlife movement corridors are passages that connect landscape patches together and are key in mitigating the effects of habitat fragmentation and to restore and maintain landscape connectivity (Hilty et al., 2012). Maintaining large-scale corridors and networks between protected areas, such as through projects like the Yellowstone to Yukon (Y2Y) project and the Appalachian Trail project, can help ensure that protected cores are connected via a system of relatively natural and barrier-free lands to promote functional connectivity (Belote et al., 2017; Conservation Corridor, 2020; Chetcuti et al., 2021). As an example, the Y2Y initiative has a vision of "an interconnected system of wild lands and waters stretching from Yellowstone to Yukon, harmonizing the needs of people with those of nature", which covers a region from the US to Canada (Aengst, 1999; Yellowstone to Yukon Conservation Initiative, 2024). The project promotes connectivity conservation across lands in multiple jurisdictions, with protected lands making up the key areas for providing a haven for native wildlife and ecological corridors enabling movement between the protected areas, along with creating a community of a variety of stakeholders to avoid patchwork management of different land types and ownership (Aengst, 1999; Chester, 2015; Lemieux et al., 2021).

An example of a corridor within a larger ecological network is the Cascade Corridor in Banff National Park (Figure 2.2.4). There is a severe restriction on wildlife movement through the local area due to increasing human activity in the Town of Banff, the Trans-Canada Highway and the Canadian Pacific Railway, and so human structures in the Cascade Corridor were removed and use of the roads and airstrip were restricted to restore the natural montane habitat and encourage the movement of large carnivores through the ecosystem (Parks Canada, 2017a). This corridor forms part of the Y2Y region.

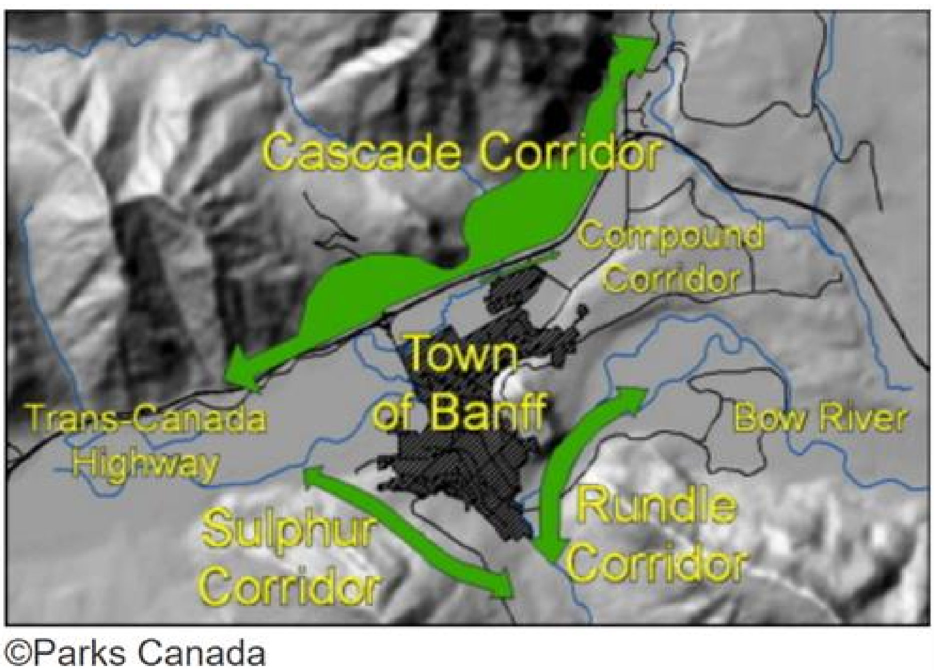


Figure 2.2.4 *- A map showing Cascade Corridor in the Bow Valley, Banff National Park (Parks Canada, 2017a).*

In fragmented landscapes, conservation is often focused on large, unfragmented habitats and adjacent ecological corridors, but the value of small habitat patches (stepping-stones) in complementing large patches should not be overlooked (Saura et al., 2014; Herrera et al., 2017; Lynch, 2019). For some species, the landscape matrix may not be particularly inhospitable to move through, thus small "stepping-stone" refuges between larger habitat patches will greatly improve these species' functional connectivity (Lynch, 2019; Stewart et al., 2019). In addition, stepping-stones may be particularly effective at maintaining functional connectivity in urban areas, where small habitat patches do not require large commitments of land or resources (Lynch, 2019). Stepping-stones improve landscape connectivity most efficiently if they have sufficient size to be of value in animal movement, and as part of a planned ecological network, such as Y2Y, rather than on an ad-hoc basis (Saura et al., 2014; Herrera et al., 2017).

Part of Parks Canada's Guiding Principles is that the complementary use and management of adjacent lands to national parks should be pursued by government and non-government agencies at several levels to maintain the integrity of ecosystems and to encourage sustainable development near to parks (Benidickson, 2011). This can be done by creating an ecological network for conservation, and by putting into practice concepts such as ecological corridors (Dietz et al., 2021; Lemieux et al., 2021). Therefore, since improving connectivity in national parks and their surrounding areas is of notable importance to the Parks Canada Agency, it is imperative that fragmentation as an ecological indicator is more closely inspected across all national parks and at a variety of temporal scales. Despite fragmentation not currently being considered by Parks Canada as the most cost-effective indicator in all national parks, this study considers all parks in the National Park System to make effective comparisons between them.

2.2.5 Changing approaches to management – tourism and early fragmentation of the wilderness

Over the 135 years of Canadian national park history, there have been huge changes in the management approaches of the protected areas. The dramatic shift from tourism and the exploitation of resources to the acknowledgement of biodiversity, habitat restoration, and eventually ecological integrity have been significant. However, historically, national parks policy in terms of both designation and management has been mostly centrist in its origins and applications, considering both the economic benefits of tourism and resource extraction, and conserving wildlife (Brown-John, 2006).

Banff National Park was the first national park to be created in Canada in 1885 as a tourist attraction, however the regulations it was created under –the Dominion Lands Act -lacked strength to protect its natural and cultural heritage (Historic Places, 2020). In general, the rationale for Canada's first parks was driven by colonial economic development and tourist dollars, along with the expansion of the Canadian Pacific Railway (Binnema & Niemi, 2006; McNamee, 2010). The landscapes in these early National Parks were advertised as untouched wilderness areas, but even before park designation they had been affected by extractive activities such as lumbering and mining (Scace, 1968; Fluker, 2010), along with the significant fragmenting elements of the new railway (Dearden & Berg, 1993). In addition, in keeping with the "untouched wilderness" ideal, Indigenous inhabitants in park boundaries were forcefully removed from their homelands so that tourists could enjoy "pristine" areas (Cronon, 1995; Binnema & Niemi, 2006; Parlee et al., 2012). In current times, it can be argued that this process of removing Indigenous communities from areas wanted for parks continues, but disguised as part of the legal system. Despite Parks Canada's designation of some areas as National Park Reserves, once an area is subject to one type of management regime, such as that of a national park, the land is effectively withdrawn from becoming a total indigenous claim (Lawson, 1987; Martin 2016). Although Indigenous representative management boards may be established for the co-management of NPRs, not all co-management models have been successful in giving equal influence to Indigenous communities in comparison to other stakeholders (Dearden & Berg, 1993; Murray 2010; Martin 2016). As such, this study places focus on the effects of settler infrastructure development on landscape fragmentation.

There was no real policy direction for parks and forest reserves until the National Parks System was officially created under the Dominion Forest Reserves and Parks Act (1911), where "wise" use of the recently designated forest reserves could be made under a mandate of conservation vs commercialization, allowing extractions of resources as well as tourism (Searle, 2000; Brown-John, 2006; McNamee 2010). This was known as the dual mandate and was possible due to language in the 1911 Act that was open to interpretation as to the purpose of the parks (Dearden & Berg, 1993). Between 1914 and 1930, thirteen new parks were dedicated to protect scenery and wildlife. However, the Parks Branch (in various forms of government) remained occupied by the parks' tourism potential to the white middle-class (Mortimer-Sandilands, 2009). Under these policies, there were improved visitor accommodations and attractions, the construction of roads and trails, and the removal of predators in the ecosystems, leading the way for increased fragmentation in the mountain parks and an upset ecological balance (Searle, 2000; Mortimer-Sandilands, 2009). For example, in Banff alone, twenty years of development from 1914 led to the Banff Springs Golf Course, Mount Rundle Campground and

Banff Recreation Ground, the Banff Airfield and the Mount Norquay, Lake Louise and Sunshine ski areas (Dearden & Berg, 1993). Also of significant note is the early construction of the National Pacific Railway and Trans-Canada Highway through the mountain parks, which both still have significant effects on landscape connectivity today, as demonstrated by the Cascade Corridor in figure 2.2.4 (Brown-John, 2006; Warner, 2008).

Following the main dual mandate era came the Canada National Parks Act of 1930, which strengthened protections for natural and cultural heritage (Historic Places, 2020). This was also the first legislation that used the language "unimpaired for future generations", although since its establishment the understanding of this phrase has evolved in conjunction with the emergence of ecological integrity as a management standard, and in relation to supposed increased acceptance of Indigenous land use activities within park boundaries (Lawson, 1987; Benidickson, 2011). Since this time, Parks Canada has mostly maintained that a dual mandate of both ecological conservation and visitor use has never existed. However, despite conservationist language in the legislation, there was a post-war boom in resource extraction even within the parks, and recreational over-use, damaging park ecosystems until the beginning of Canada's wildlife preservation movement in the 1960s (Searle, 2000).

2.2.6 Changing approaches to management – environmental concerns and striking a balance

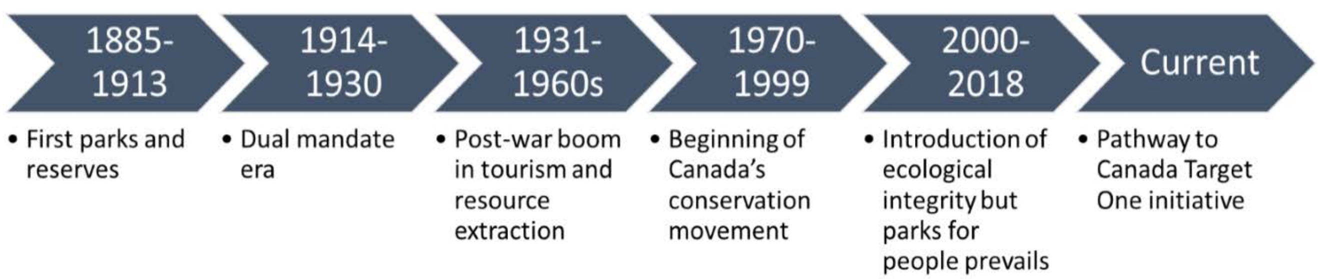
In the mid-19th century, there were generally three main expectations of national parks in the public's understanding: scenic beauty, recreation opportunities and wildlife preservation (Neufeld & Campbell, 2011). A rapid increase in vehicle traffic and road accessibility to many national parks caused a tourism explosion, (Colpitts, 2012) and both legal and illegal mining and logging activity, particularly in the North around areas that were being considered for national park status, was occurring due to park mismanagement (Lothian, 1974; Timoney, 1996; Neufeld & Campbell, 2011). As highway tourism brought more visitors to parks, there was increased demand for facilities, creating a snowball effect of more roads and development (Mortimer-Sandilands, 2009; Taylor & Campbell, 2011). Parks were marketed as playgrounds in postwar promotion, and development was seen as being positive for economic growth (Saari, 2015).

During Canada's revival of the wildlife preservation movement from the 60s onwards, environmentalists challenged big business entrepreneurs and the government to better protect existing parks, preserve "wilderness" and create new parks (Dearden & Berg, 1993; Mortimer-Sandilands, 2009). A variety of new conservation groups formed, such as the National and Provincial Parks Association of Canada (now CPAWS) and Canadian Audubon Society (now Nature Canada), due to the cultural revival of the environmental movement in North America (McNamee, 2010). This contributed to the amendment of the National Parks Act in 1988, which was mostly pro-environmental, along with the Canadian Environmental Protection Act being passed into law (Dearden & Berg, 1993; ECCC, 2021). The 1988 National Parks act identified ecological integrity as formal policy (Brown-John, 2006)

Despite these policy changes, by the time of the 1997 State of Parks report, it was determined that almost all national parks were experiencing significant threats to biodiversity and damage to ecological integrity (Benidickson, 2011), due to a lack of formalization of the obligation to pursue ecological integrity, and a reduction in federal support to the Parks Canada Agency (Dearden & Berg, 1993; Brown-John, 2006; Mortimer-Sandilands, 2009). The 2000 amendment of the Canada National Parks Act formally defines ecological integrity by the following: "an ecosystem has integrity when it is deemed characteristic for its natural region, including the composition and abundance of native species and biological communities, rates of change and supporting processes", and in 2001 the federal government legislated ecological integrity as the priority of management in national parks (Fluker, 2010).

However, preservation policy was having insignificant effect on curbing the prevailing "parks for people" ideology, and in recent years there has often been a growing disconnect between national parks legislation and Parks Canada's programs (Fluker, 2010; CPAWS, 2019). As demonstrated by Searle (2000) in "Phantom Parks", despite policy pushing for ecological integrity, the parks were still under pressure from heavy anthropogenic use, mainly tourism, causing severe landscape fragmentation in many parks. In the highly developed south of Canada, the most pervasive form of development and fragmentation was roads (Leroux & Kerr, 2013), and inherent to the southern parks is economic development and business opportunities relating to tourism (Orr, 2014). Sixteen years after the amendment to the Canada National Parks Act, a similar pattern was observed by the independent CPAWS report of 2016: it was noted that there had been a significant shift in Parks Canada's management approach away from preservation towards a more touristic and marketing focused approach and over-investment in visitor facilities, which put ecosystems and biodiversity in parks at risk (CPAWS, 2016; Weber et al., 2019). However, by the 2019 report, it was recorded that momentum from the Pathway to Canada Target One Initiative and international targets had resulted in an increase of conservation work across the country (CPAWS, 2019). Parks Canada's current national zoning framework plays a part in attempts to balance tourism with protection, ranging from the high visitor use Zone V – Park Services, to areas of undisturbed wilderness in Zone I – Special Preservation (Thede, 2011).

Overall, the changing styles of management across the entire National Park System can be roughly placed into six time-steps:



The running theme of heavy anthropogenic pressures on parks, even despite preservation-focused and ecological integrity-focused policies, demonstrates that it is essential to keep monitoring ecological indicators such as fragmentation over various time-steps and to identify the trends. It is not enough to take the Parks legislation at face value and assume that the ecosystems in and around park boundaries are suitably protected. Overall it can be said that "the Canada National Parks Act definition of Ecological Integrity is[...]a sophisticated model of the late nineteenth century socially constructed wilderness, combined with late twentieth century ecosystem science" (Fluker, 2010:9).

2.2.7 Concluding remarks

Conservation actions are required to reduce fragmentation, improve connectivity and restore or maintain habitat quality for species, including the establishment of new protected areas and the enlargement of current national parks, modifying or removing resource extraction practices, the establishment of wildlife corridors, and the prevention of heavy anthropogenic use of national parks, tourism-related or otherwise (Crist et al., 2005; Whittington et al., 2019). Given the strong negative effects on biodiversity by increasing fragmentation, and the need for maintaining and restoring landscape connectivity particularly for key species in Canada's national parks, a comprehensive study of fragmentation across the National Parks System is necessary to ensure that Canada's rich biodiversity has enough habitat to maintain itself. Explicit monitoring and reporting on the degree of fragmentation will be essential for identifying threats and changes in trends and evaluating compliance with legislation.

3 - Paper manuscript: How effective have national parks in Canada been since their designation at preventing landscape fragmentation?

3.1 Introduction

Parks Canada's mandate to maintain or restore the ecological integrity of its national park system (NPS) warrants an analysis of landscape fragmentation and its change over time (Canadian National Parks Act, 2000; Fluker 2010). Ecological integrity with respect to a park indicates that it is in a condition that is characteristic of its natural unimpaired state and is likely to persist. Any increase in human activity in Canadian national parks and park reserves is likely to affect their landscape structure and biodiversity, and will provide an indication of impacts on the integrity of the native ecosystems of parks and their surrounding areas (Soverel et al., 2010). 'Landscape fragmentation' is a species-agnostic term, rather than the more specific 'habitat fragmentation', making it relevant for use across a variety of ecosystems such as those in the NPS. Currently, fragmentation is loosely monitored through indicators in the agency's ecological integrity monitoring framework (EIM) and through other park-specific projects.

Connectivity between natural landscapes is increasingly becoming the focus of Canada's conservation goals (Pathway to Canada Target One, 2021; Parks Canada, 2023f). Habitat loss and the break-up of habitat patches change landscape configuration, creating barriers for wildlife movement and reducing a landscape's connectivity (Goodwin & Fahrig, 2002). In addition, with unfragmented parts of landscape becoming smaller, habitat extent and quality is reduced, and wildlife populations are subdivided into smaller, more vulnerable fractions (Jaeger et al., 2008; Roch & Jaeger, 2014). Canadian national parks are at risk of (a) becoming islands of natural habitat that are disconnected from their greater ecosystems, and (b) becoming further fragmented from stressors both inside and outside park boundaries (MacArthur & Wilson, 1967; Rivard et al., 2000; Parks Canada, 2019a).

Over the history of Parks Canada, there is a running theme of heavy anthropogenic pressures on its protected areas, despite a turn towards conservation-focused policies. The past 90 years since the National Parks Act have marked a struggle by Parks Canada to strike a balance between the economic benefits of tourism and resource extraction, and conserving wildlife and intact landscapes (Brown-John, 2006). This struggle continues today as demonstrated by the 2022 Ecological Integrity of National Parks report, according to which 35% of parks were in fair or poor condition and 21% of parks had declining ecological integrity (ECCC). Monitoring fragmentation and the rate at which it changes in parks is therefore important to ensure that the composition of native ecosystems, rates of change, and supporting processes are characteristic of the parks' natural regions.

This study measured the fragmentation of Canadian national parks and associated control areas at key time-steps throughout their history, including before their designation or conservation in 1930 with the *Canadian National Parks Act* up to 2020 using the Effective Mesh Size metric (Jaeger, 2000). A Progressive-Change Before-After Control-Impact Paired Series design then served to evaluate the divergence of landscape fragmentation of national parks and park reserves in Canada from associated control areas since their designation (Thiault et al., 2017). To our knowledge, there has not been an analysis of fragmentation that either covers this long timeframe of over 90 years of Parks Canada history, nor the majority of Canada's NPS enabling comparisons between parks.

A set of hypotheses drives this study. The first, and main hypothesis relating to anthropogenic influences on parks is that *a park is likely to be more fragmented than other regions if it has higher visitor numbers, a longer history of human influences in the parks and a greater number of transport links leading into the park*. A general prediction would be that Canada's federally-protected areas are successful in maintaining ecological integrity, thus fragmentation levels have not increased. Another hypothesis is that the *fragmentation levels and rates in the protected areas are much lower than in the control areas.* Landscape fragmentation is caused by urbanization, roads, and related anthropogenic infrastructure and an increase in agricultural lands. However, tourism is a key economic driver and inherent stressor on Canada's national parks (Bath & Enck, 2003; Benidickson, 2011), and the related human development and activities are likely to have a fragmenting effect on the so-called 'natural' landscapes of federally-protected areas (CPAWS, 2019). Lastly, it can be hypothesized that *the longer a national park has been protected, it will have more landscape connectivity than parks that have been protected for less time.*

Accordingly, four research questions are addressed:

- How do fragmentation levels and rates of change compare before and after the establishment of National Parks in Canada?
- How do fragmentation levels and rates compare between the protected areas and the nearby non-protected (control) areas?
- How do the lengths of time of protection of national parks relate to their levels and rates of fragmentation?
- How does the remoteness of parks from human activities affect their fragmentation levels?

3.2 Methods

This project assessed 43 National Parks and National Park Reserves that are under the jurisdiction of Parks Canada (Table 3.2.1a). This is a wide selection encompassing the majority of Parks Canada's protected areas, in order to represent and compare between Canada's natural regions. Sable Island National Park Reserve was not included due to its small size ($34km^2$) and its unique, extremely remote nature. Thousand Islands National Park, Georgian Bay Islands National Park and Mingan Archipelago National Park Reserve were also excluded due to the lack of a suitable control area – there is no other group of islands in the same region that are unprotected. A major goal of Parks Canada is to establish a system of national parks that represents all of these natural regions, a goal, which as of 2021, is "just over 60% completed" (Parks Canada, 2021d).

Table 3.2.1a*- Canadian National Parks and Park Reserves that were included in this fragmentation analysis. Marine parks and National Historic Sites were not included.*

National Parks			**National Park Reserves**
Aulavik	Jasper	Riding Mountain	Akami-uapishk^{U}-KakKasuak-Mealy Mountains
Auyuittuq	Kejimkujik	Sirmilik	Gulf Islands
Banff	Kootenay	Terra Nova	Gwaii Haanas
Bruce Peninsula	Kouchibouguac	Torngat Mountains - Tongait KakKasuangita SilakKijapvinga	Kluane
Cape Breton Highlands	La Mauricie	Tuktut Nogait	Nááts'įhch'oh
Elk Island	Mount Revelstoke	Ukkusikalik	Nahanni
Forillon	Point Pelee	Vuntut	Pacific Rim
Fundy	Prince Albert	Wapusk	Thaidene Nëné
Glacier	Prince Edward Island	Waterton Lakes	
Grasslands	Pukaskwa	Wood Buffalo	
Gros Morne	Qausuittuq	Yoho	
Ivvavik	Quttinirpaaq		

3.2.1 Park boundaries and control area selection

The reporting units include the areas within the boundaries of the National Parks and National Park Reserves, together with corresponding control areas. The control areas were selected based on their similarity to their respective protected areas before their designation as National Parks and Park Reserves. The control area selection considered the following factors: reporting unit area extent, proximity to the park, ecoprovinces of the parks and control areas (Agriculture and Agri-food Canada, 2020), waterbody amounts, and overall land cover, in order to account for the potential issue of natural change over time due to the development of ecosystem processes and communities in both park and control areas (Smith et al., 1993). Each park and control area were given a similarity score out of 8, based on the factor scores as shown in Table 3.2.1b for full transparency of park and control site matching. These factors were chosen for their ability to indicate the coherence of a pair of sites if there was an absence of a treatment to one site. In this study, the treatment was the designation of park status to one of the sites in the pair (Osenberg et al., 2006; Thiault et al., 2017).

Table 3.2.1b - Factors considered in similarity scores for the association of control areas to parks. Full table provided in appendix.

Factor	Reason for inclusion	Similarity score	Example – Nááts'įhch'oh (Figure 3.2.1)	Data sources
Area extent	Must be the same as park area for fragmentation calculation - particularly for the CUT m_{eff} procedure.	1 – same as park	Same as park	National Parks and National Park Reserves of Canada Legislative Boundaries shapefiles, Natural Resources Canada (2020a)
Proximity to park, at closest point (km)	A control area must be near enough to its paired park site to reduce spatial variability between the two sites.	1 – under 100km, 0 – over 100km	12 km	Natural Resources Canada (2020a)
Ecoprovince	Suggests similar regional characteristics across the ecoprovince, such as climate, elevation and geology.	2 – all same, 1 – one ecoprovince in common, 0 - different	Mackenzie-Selwyn Mountains	Terrestrial Ecoprovinces of Canada feature dataset, Agriculture & Agrifood Canada (2020b)
Waterbody area comparison (% difference)	An important covariate for land mammals for which a waterbody could seriously impede movement.	2 – under 5%, 1 – under 10%, 0 – over 10%	0.25%	Lakes, Rivers and Glaciers in Canada – CanVec Series – Hydrographic Features shapefiles, Natural Resources Canada (2020b)
Overall land cover	Infers a similarity of habitat types across the region, and can influence the format of anthropogenic land use.	2 – same land cover descriptions, 1 – some similar, 0 - different	Sparsely Vegetated/Barren land. Tundra: treeless arctic and alpine vegetation. Coniferous Forest	Land Cover by Ecoprovince shapefiles, Agriculture & Agrifood Canada (2020b)
Total			8/8	

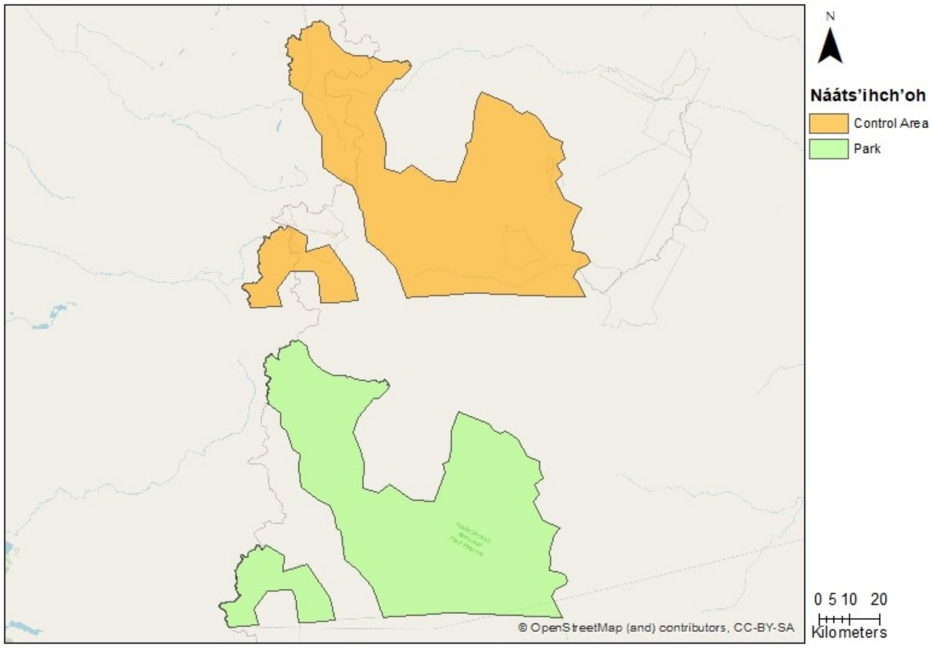


Figure 3.2.1- A map demonstrating the location of the control area for Náátsʼįhchʼoh NPR in comparison to the park reserve

Study designs in ecology lacking appropriate control sites are often considered to be less robust (Manly, 2008; Christie et al., 2019; Wauchope et al., 2022). This study uses a BACIPS design, as described in section 3.2.5, which reduces the effects of nonrandom spatiotemporal variation by pairing a control site and an impact site based on their coherence geographically and over time (Stewart-Oaten et al., 1986; Manly, 2009; Rassweiler et al., 2021). Pairing the impact sites suitably with corresponding control sites reduces the effects of any pre-existing differences in the Before period between the sites in each pair. To avoid spatial pseudoreplication, the control sites needed to be far enough from the protected areas to avoid any spillover influences from enforcement of park protection, but close enough to still be influenced by the same regional natural phenomena (Stewart-Oaten et al., 1986; Osenberg et al., 2006). In the majority of parks, this requirement was addressed by assessing each pair's ecoprovince and land cover, and by considering a 10km CBC buffer zone around each site.

The process of selecting control areas for each park involved the use of GIS polygons of the Parks and Park Reserves. Copies of these polygons were moved to the control area locations, thus creating the new control area layers. Basemaps from OpenStreetMap (OpenStreetMap, 2021), topographic maps from the Canada Centre for Mapping (Natural Resources Canada) and vector hydrographic and land feature data from CanVec were used to determine the appropriate places for the control areas, as demonstrated in Table 3.2.1b.

3.2.2 Vector data and historical map data collection

GIS datasets (both vector and paper maps) were evaluated regarding their suitability for quantifying fragmentation in National Parks and their respective control areas. Five suitability criteria were considered to select the datasets to use:

1. Time-step updates that correlate to the timeline of changing approaches in Canada's National Parks (Table 3.2.2b);
2. Complete area coverage of the national parks and control areas;
3. Definitions of fragmenting elements remain as consistent and unambiguous as possible across the national parks and control areas, and over time;
4. Resolution remains reasonably consistent over time, with consideration that this may not be possible for historical data;
5. Contains information about all human influences in the national parks and control areas that are considered in fragmentation geometries 1 & 2 (Table 3.2.2a).

Table 3.2.2a *- Data suitability table.*

Data	Suitable time-step?	Complete coverage of NPs and CAs?	Consistent definitions of fragmenting elements?	Consistent resolution?	Inclusion of data for all FGs?
CanVec Series	2021	Yes	Yes	Yes	Yes
Open Database of Buildings	2020	Yes	Yes	Yes	NA[1]
Annual Crop Inventory	2009-2020	Yes	Yes	Yes	NA
CanVec Series	2013	Yes	Yes	Yes	Yes
CanMatrix Series	1944-2012	Yes	Yes	Yes	Yes
Surveys and Mapping Branch. Energy, Mines and Resources Canada	1954-1987	Yes	Yes	Yes	Yes
Department of the Interior	1921-1931	No[2]	Yes	Yes	Yes

Most vector data were downloaded from open-source Topographic Data of Canada – CanVec Series (CanVec Series, 2021). This vector series contains over 60 topographic features in 8 themes: transport, administrative, hydrological, land, manmade, elevation, resource management and toponymic features. Open Street Map was used to corroborate these vector data. Open Street Map has the benefit of providing open access to anyone, enabling a "diverse variety of individuals, communities and organisations" to contribute to the data (OSMfoundation, 2021). This provides a remote form of ground-truthing, because it is primary data from sources at the locations in question, as travel to most national parks was beyond the scope of this study. Further information about buildings and contours were extracted CanVec elevation contours and the Open Database of Buildings (Statistics Canada, 2019). Archived vector data for earlier points in time were downloaded from open-source Topographic Data of Canada – CanVec 1:50,000 (release 2013) (CanVec Series, 2013). These data contain the same 8

[1] Data denoted as NA is not needed in any or all fragmentation geometries related to human impacts (i.e. FGs 1 & 2)
[2] Gaps in geographic coverage filled with alternative maps – further information in appendix

themes of topographic features as the up-to-date vector data, and were downloaded in 1:50,000 squares that covered all protected areas and control areas.

Agriculture data from 2009 to 2020 were converted from raster to vector data using AAFC's open-source Annual Crop Inventory (2020) and the Land Cover for Agricultural Regions of Canada (2000) Agricultural area was extracted by attributes from the complete land cover data, and ArcMap's *Raster to Polygon* tool was used to convert the overall agricultural land cover into vector data.

Historical map data pre-2000 were collected from a wide variety of sources. A significant source of data for the years 1970-1999 was the open-source group of Digital Topographic Raster Maps from Natural Resources Canada (CanMatrix Series, 2008), which were available in georeferenced TIFFs. Maps for earlier time-steps were generally more difficult to locate, particularly because of the closure of Government of Canada storerooms due to the Covid-19 pandemic. Therefore, many of these older maps were retrieved from university library collections, namely the University of Toronto, the University of Alberta William C. Wonders Map collection, the UBC Koerner Library, McMaster University Map collections, and the University of Calgary Historical Maps Collection. In 2022, the open-source Borealis Historical NTS collection expanded to include topographic maps of many more rural areas in Canada from 1948 to the early 2000s. This filled in the majority of the data gaps for the 1931-1969 post-war time-step using 1:25,000 squares. Finally, some maps from the 1914-1930 dual mandate era were downloaded from the open-source Canadiana Heritage Reels (Canadian Research Knowledge Network, 2021), the Sectional Maps collection at the University of Calgary (SANDS, n.d.). Of these historical maps, the majority were government-standard topographic maps from various interior departments. Others were tourist maps from Parks Canada and provincial governments. Additionally, it was important to consider name changes over time of some protected areas in order to find accurate historical data; for example, Banff NP was originally known as Rocky Mountains NP and Ivvavik NP was originally known as Northern Yukon NP.

__Table 3.2.2b__ – Time-steps based on Parks Canada's changing approaches towards park protection.

Years	Description	Parks created during each era
1885-1913	First parks	Banff, Glacier, Yoho, Waterton Lakes, Jasper
1914-1930	Dual mandate era	Elk Island, Mount Revelstoke, Point Pelee, Kootenay, Wood Buffalo, Prince Albert
1931-1960s	Post-war boom in tourism and resource extraction	Riding Mountain, Cape Breton, PEI, Fundy, Terra Nova, Kejimjujik, Kouchibouguac
1970-1999	Beginning of Canada's wildlife preservation movement	Forillon, La Mauricie, Pacific Rim, Auyittuq, Kluane, Nahanni, Gros Morne, Pukaskwa, Grasslands, Ivvavik, Bruce Peninsula, Gwaii Haanas, Quttinirpaaq, Aulavik, Vuntut, Wapusk, Tuktuk Nogait
2000-2018	Introduction of ecological integrity but 'parks for people' prevails	Sirmilik, Gulf Islands, Ukkusiksalik, Torngat Mountains - Tongait KakKasuangita SilakKijapvinga, Náátsʼįhchʼoh, Akami-Uapishk^{u}-KakKasuak-Mealy Mountains, Qausuittuq
Current	Pathway to Canada Target One	Thaidene Nëné

For the oldest parks and corresponding control areas in the Canadian National Parks System, fragmentation levels were assessed for up to five points in time. Line graphs were created for each individual park-control pair showing years since park designation against m_{eff}, along with scatterplots of the BACI change for each park-control pair across the NPS. This enabled evaluation of how the sizes and time since protection of the parks relate to their levels of fragmentation and rates of change. Due to a lack of reliable data, the first time-step of 1885-1913 was omitted, and parks that were designated in this time frame were assessed as part of the 1914-1930 time-step. Common map data from this earliest time period include hand-drawn sketches of the areas, maps only including a few anthropogenic features, and/or maps that are too spatially-inaccurate to be georeferenced.

To map the changes in landscape fragmentation along these time-steps, some of the older datasets used were in the format of hard-copy maps or image files with no georeferencing treatment. Therefore, a digitization method was required using ArcMap 10.7 (Figure 3.2.2). Hard-copy maps were scanned and/or downloaded, and georeferenced (when required) using consistent topological markers as control points, such as mountain peaks and lakeshores. Once enough control points across the entire park locality were created (Chapman & Wieczorek, 2020), the transformation *Adjust* was used to determine the correct geographical locations for the cells in the scanned map raster. This transformation was chosen because it uses both a global least-squares fitting algorithm and local accuracy, and is based on at least three control points; most georeferenced scanned maps had at least 8 control points (Esri, n.d.). The georeferenced scanned maps were then digitized into vector layers using the same element identifiers as the more recent data, such as roads, trails, and built-up urban areas (Fenech et al., 2005). These polygon vector layers were then incorporated into fragmentation geometries and used for the calculation of the Effective Mesh Size of each reporting unit over the various time-steps, as described in section 3.2.3 below.

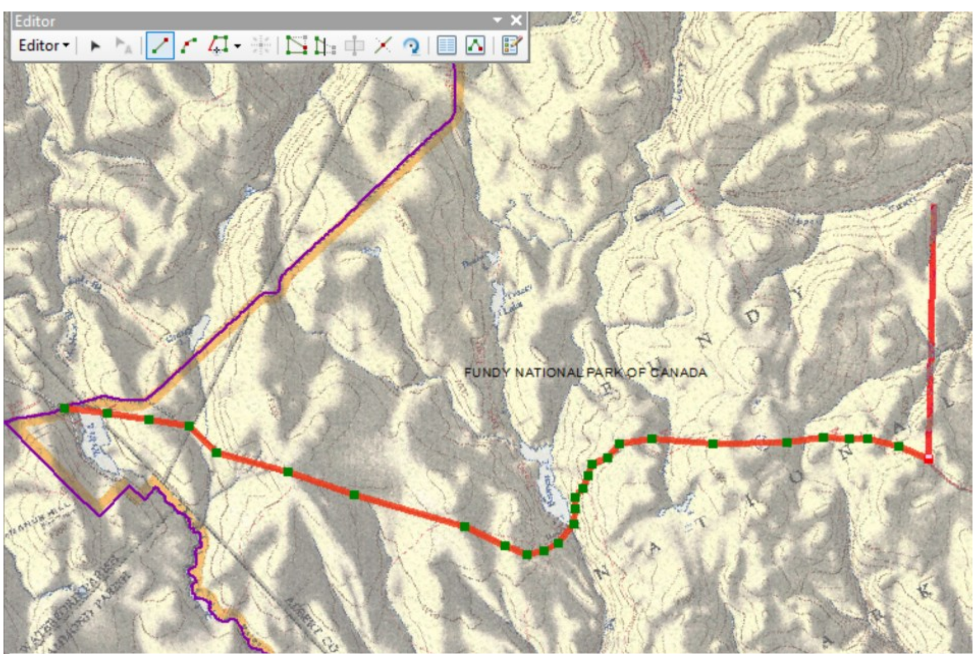


Figure 3.2.2 *– Example of digitizing a highway in Fundy National Park from a 1962 map*

The regions in this study were created by merging related terrestrial ecozones into ecoregions (Statistics Canada, 2018). The parks in these regions were assessed as subgroups to provide further context to how the remoteness of parks from human activities affect fragmentation levels inside and outside park boundaries.

- Arctic: Arctic Cordillera, Northern Arctic, Southern Arctic
- Taiga: Taiga Plains, Taiga Shield
- Pacific Maritime
- Cordilleras: Boreal, Taiga & Montane Cordilleras
- Atlantic Maritime & Great Lakes: Atlantic Maritime, Mixedwood Plains
- Prairies: Prairies, Boreal Plains
- Hudson: Boreal Shield, Hudson plains

3.2.3 Creation of fragmentation geometries

To quantify landscape fragmentation for this study, it was necessary to identify the landscape elements that affect the fragmentation of ecological processes and animal movement in national parks (Girvetz et al., 2008; Jaeger et al., 2008). Anthropogenic fragmentation may include roads, railroads, urban and industrial areas, and agricultural areas. However, some natural elements could also be considered as fragmenting elements, such as high elevation areas (e.g., when covered in ice, snow and/or scree), watercourses, and permanently flooded areas. In this landscape fragmentation analysis, a fragmentation geometry specifies which elements of fragmentation, i.e., barriers to movement, have been considered (Roch & Jaeger, 2014).

An array of fragmentation geometries (FGs) was created and used for a detailed analysis of landscape fragmentation, instead of one single FG. This allows for a variety of interpretations of landscape fragmentation (Jaeger et al., 2008), depending on the types of barriers that are relevant for different groups of species. This considers habitat specialists that are restricted to one type of habitat through FG4 and FG5, and habitat generalists that live in a greater variety of habitats through FG1-3 (Roch & Jaeger, 2014). Anthropogenic fragmenting elements in Canadian National Parks include: paved and unpaved roads, railroads, trails, campgrounds, golf courses and ski areas, among other tourist-centric facilities. Natural fragmenting elements include mountains, watercourses, and lakes.

Table 3.2.3a *– Definition of the fragmentation geometries used in this study*

ID	Features included
FG1	Only tourist elements, paved roads and built up areas.
FG2	All anthropogenic barriers: including those in FG1 and: productive agricultural land, railroads, unpaved roads, and industrial areas.
FG3	In addition to FG2, watercourses, waterbodies, glaciers and ice fields, and other natural fragmenting elements included.
FG4	In addition to FG3, high elevation is included with peaks 250m above alpine tundra counted as fragmenting elements (Trant, 2020)
FG5	As in FG4, with hydrological areas excluded from the study area

FG1 contains the strongest barriers to species, that would impede the movement of most habitat generalists. FG2 contains almost all human-made barriers, to demonstrate the effect of all anthropogenic activities in national parks. FG3 considers hydrological features, as these are often barriers to terrestrial animal movement. In some cases, FGs without waterbodies and watercourses can also be considered as seasonal fragmentation geometries in some northern parks - where large bodies of water freeze over in winter and provide connectivity for animal movement. FG4 represents the maximum degree of fragmentation by considering all possible natural and anthropogenic barriers, including high elevation. Finally, FG5 excludes the hydrological areas and only considers terrestrial land, thus excluding the additional barriers that are not part any of the terrestrial animals' possible movement space within or from the reporting units (Jaeger et al., 2007).

To include the barrier strength of each fragmenting element and a disturbance zone around them, buffers of varying widths were added to the fragmenting elements (Table 3.2.3b). These buffered elements were then merged according to each FG, to create a single polygon layer covering both the park and control area, and the 10km buffer surrounding them. This created pre-processed patch layers to be used in the effective mesh size calculations. For the patch-size maps, the polygon layer for FG2 was erased from the reporting unit layer to create a layer of landscape patches.

Table 3.2.3b -Fragmenting elements and their inclusion in fragmentation geometries

Feature	.shp type	FG1	FG2	FG3	FG4	FG5	Buffer (m)	Reason
Campsites/picnic areas	Pt						300	sqrt of 95,000 m^2 - mean of campground and picnic sites polygons across Canada (CanVec, 2021)
Ski hills/golf courses	Ply						200	sqrt of 40000 m^2 - mean of golf courses polygons (CanVec, 2021)
Boating features	Pt						1	Low due to the small effect on terrestrial connectivity in this study - manmade hydrological features affect marine connectivity more
Airfields/runways	Ply							
Buildings (incl. Towers)	Pt						15	sqrt of 220 m^2 - mean of building footprints in the Open Database of Buildings (Statistics Canada, 2019)
Built-up areas	Ply							
Foot/cycle trail	Ln						2[3]	Minimum width is 3.5 m clearing for a multi-use clearing (Trans Canada Trail Committee, 2002)
Single lane road	Ln						11	5m buffer + 6m of average 2 lanes width in Canada (Girvetz et al., 2008; EEA & FOEN 2011; NCCHPP, 2014)
Highway – 1 lane	Ln						16	10 m buffer + 6 m of average 2 lanes width in Canada (Girvetz et al., 2008; NCCHPP, 2014)
Unpaved road	Ln						10	Leipus et al. (2010)
Railway	Ln						30	CPR right of way is 15.25 m from centre of railway both sides. Vegetation is removed from all right of way (Girvetz et al., 2008; CPR, 2022)
Mining/oil/gas industry	Ply							
Productive agricultural land	Ply							
Watercourses (permanent)	Ln						5	Lee et al. (2004)
Waterbodies	Ply					Excluded		
Glaciers/Ice fields	Ply							
High elevation	Ply							250 m above alpine tundra

[3] Buffer applied on either side of linear features.

3.2.4 Fragmentation analysis

The effective mesh size method (m_{eff}) was used for measuring fragmentation (Jaeger, 2000; Moser et al. 2007). It has favourable properties such as: being suitable for comparing the fragmentation of regions of varying total areas and different barrier strengths; being unaffected by the inclusion or exclusion of very small patches; and describing the structure of a barrier network in an ecologically meaningful way (Jaeger, 2000, 2002; Roch & Jaeger, 2014). Due to the wide variety of reporting unit sizes, barrier structure, and species movement abilities across the NPS, the effective mesh size is suitable for this study. Studying the effective mesh size of parks and control areas over time provides a comprehensive overview of levels and rates of fragmentation over time, through the use of one accessible landscape metric. The metric is based on the probability that any two points chosen randomly in an area are connected, without a separation by barriers to movement. It indicates the degree to which animals can move freely in the landscape without encountering such barriers, by multiplying the probability of connection of any two points by the total area of the reporting unit (Jaeger, 2000; 2008). Overall, the smaller the effective mesh size, the more fragmented a landscape is. It can also be extended to include the permeability of fragmenting elements for species moving through the landscape (Jaeger et al., 2008; Deslauriers et al. 2018; Spanowicz & Jaeger, 2019; Freeman-Cole & Jaeger, in prep.).

The metric employs an equation that results in one simple, intuitive value that meaningfully describes the spatial structure of the landscape (Jaeger et al., 2007), therefore making the metric accessible to a variety of stakeholders in the land and the general public. A major mathematical benefit of the effective mesh size is that it is area-proportionately additive and intensive – meaning that each reporting unit contributes to a group of reporting units proportionately to its size, no matter its internal spatial structure (Jaeger, 2000; Moser et al., 2007). This property was particularly useful for parks and their control areas that do not have contiguous boundaries, such as Grasslands, Gulf Islands, and Gwaii Haanas.

There are two versions of the effective mesh size metric that were used: the *Cutting-Out* (CUT procedure), in which only land within the reporting unit boundaries was considered; and the *Cross-Boundary-Connections* (CBC procedure), which has the advantage of removing any bias that would be caused by non-physical boundaries of the reporting units (Moser et al., 2007; Jaeger et al., 2008).

The formula to calculate m_{eff} for a given reporting unit according to the CUT procedure is:

$$m_{\mathrm{eff}}^{\mathrm{CUT}} = \frac{1}{A_{\mathrm{total}}} \sum_{i=1}^{n} A_i^2$$

where n = number of patches inside the reporting unit; A_i = sizes of the n patches ($i = 1, \ldots, n$); and A_{total} = total area of the reporting unit within its boundaries. The value of m_{eff} varies between 0 and the total area of the reporting unit (A_{total}).

The formula to calculate m_{eff} for a given reporting unit according to the CBC procedure is:

$$m_{\mathrm{eff}}^{\mathrm{CBC}} = \frac{1}{A_{\mathrm{total}}} \sum_{i=1}^{n} A_i \cdot A_i^{\mathrm{cmpl}}$$

where n = the number of patches, A_i = size of patch i inside the boundaries of the reporting unit; A_i^{cmpl}= the area of the complete patch that A_i is a part of, including the area on the other side of the boundaries of the reporting unit up to the physical barriers of the patch; and A_{total} = the total area of the reporting unit (Moser et al., 2007; Girvetz et al., 2008).

For the CBC method, a buffer zone of 10km was applied to capture patches that extend over the NP or NPR non-physical boundaries (i.e., there is no fence demarcating the protected area and surrounding locality). Since this is a landscape-scale study and is not species-specific, the same 10km buffer size was used for every park and control area (Lawrence et al., 2021). This choice of buffer was inspired by the daily movement of four subspecies of caribou (*Rangifer tarandus*) found in Canada: boreal, southern mountain, Peary, and barren-ground. Caribou movement was considered here because the four subspecies cover a range across Canada, and they are a keystone species in most of the ecosystems they are found in (CPAWS Saskatchewan, 2021; ECCC, 2021). In addition, their daily movement range is similar to that of elk and other ungulates (Rosatte, 2016), and their movement is influenced significantly by roads and other anthropogenic development (Wilson et al., 2016). To determine buffer size, the average daily movements of caribou were taken from six studies (Table 3.2.3c). This accounted for the high seasonality of their movement, where caribou cover a much greater distance in a day during migration than in the calving season – for example, the annual migration of the Porcupine caribou herd is the largest of any land animal on earth (Johnson et al., 2002; Ferguson & Elkie, 2006).

Table 3.2.3c *- CBC buffer zone creation process using literature about caribou movement*

Range of daily movement (km)	Source
2.3-9.1	Poole et al., 2010
2	Pedersen et al., 2021
0.26-0.69	Wakkinen & Sloane, 2010
2.8-18	Person et al., 2007
5.8-15.9	Gunn et al., 2013
2-11.5	Russell & Gunn, 2019
Mean of ranges: 9.13km	

The mean of these selected daily movement ranges is 9.13km. This was rounded up to 10km for the buffer size for ease and reproducibility of the m_{eff} CBC calculations.

Finally, the results from these formulae (CUT vs. CBC) were compared for each fragmentation geometry so that the effect of nearby fragmentation adjacent to the protected areas could be considered (Lawrence et al., 2021).

The m_{eff} for the parks and control areas, both for the CUT and CBC methods, were calculated using the *ZonalMetrics toolbox extension* (Wetzel et al., 2019; Walz et al., 2021) for ArcGIS Desktop 10.1 onwards and ArcGIS Pro. This toolbox extension to the original *ZonalMetrics* Python toolbox (Adamczyk & Tiede, 2017) includes a tool to calculate m_{eff}, used by inputting a reporting unit layer of uncut open space, a layer of patches as defined by their fragmentation geometries – the "Processed Input Layer" - and by selecting the calculation method to be used – either CUT or CBC. Although it is possible to use unprocessed input layers - described by Wetzel et al. (2019) and Walz et al. (2021) as "the layer that contains the natural patches" - in this study it was more efficient to pre-process the fragmentation geometries into single polygon layers from the various data sources.

3.2.5 Statistical analysis

The last step in analyzing the fragmentation history of each park and of the whole national parks system, was the evaluation of the changes in the effective mesh size along the time-steps, and between the protected areas and the control areas.

3.2.5.1 High inferential strength of BACI designs

This study uses a type of Before-After Control-Impact Paired Series (BACIPS) study design to allow for the evaluation of fragmentation on a site. The changes in the fragmenting elements are compared before designation or early on in conservation and at various time-steps after the designation of an area as a National Park or Park Reserve.

In this design, the Park sites (I - impact) are compared with undesignated control sites (C), before (B) and after (A) the park designation date and along the various time-steps, producing a final value of divergence in effective mesh size. This divergence provides a measure of the effect of the intervention on the landscape – therefore providing a measure of how fragmentation levels and rates of change compare before and after park designation in both the park and control area. A general overview of the BACI term interactions can be seen in the following:

$$\mathrm{Value} \sim \mathrm{Time} + \mathrm{BA} + \mathrm{CI} + (\mathrm{BA} \cdot \mathrm{CI}) + (\mathrm{BA} \cdot \mathrm{Time}) + (\mathrm{CI} \cdot \mathrm{Time}) + (\mathrm{BA} \cdot \mathrm{CI} \cdot \mathrm{Time})$$

The BA term is binary, as 0 in time-steps before the intervention, and 1 in time-steps after. The CI term is also binary, accounting for the impact sites and control sites. The interaction between *BA* and *CI* describes the BACI immediate change, while the interactions with *Time* describe the BACI trend change (Wauchope et al., 2021).

Overall, BACI designs have high inferential strength, particularly where a variable has changed over time (in this case the level of fragmentation) between the control and impact sites. This gives strong evidence that it is the intervention (in this case the park designation) that has caused the observed difference or change, or the intervention prevented a change that would otherwise have been likely to occur (Roedenbeck et al., 2007; Christie et al., 2019). By assessing landscape fragmentation at a number of time-steps after the intervention, it can be assessed whether park designation has caused any temporary changes in fragmentation levels that then return to similar levels of anthropogenic development as before designation, or if the effect has been more permanent on park fragmentation levels (Thiault et al., 2017). However, BACI designs can suffer from nonrandom effects based on spatiotemporal variation:

- Spatial pseudoreplication: If the control and impact sites are geographically too close, they may both be influenced by the intervention in the impact site – the park protection (Smith et al., 1993; Thiault, 2014; Kerr et al., 2019). This is accounted for here by considering a 10km buffer around each site, and geographic proximity in the pair selection process.
- Temporal pseudoreplication: A repeated measurement at one site will more likely be similar if the measurements are closer together in time (Stewart-Oaten et al., 1986; Manly, 2008). This has been accounted for in this study by calculating m_{eff} values per time-step based on different management strategies, rather than exact year, for the park and control areas (Christie et al., 2019). Any short-term effect or noise will not affect samples that are spread far enough apart over time, and complex ecological reactions that have progressive responses can be accounted for (Stewart-Oaten et al., 1986; Thiault et al., 2017).

Although a single Before sample is often insufficient for model development in BACIPS designs (Osenberg et al., 2011), it is somewhat accounted for in the approach used here that focuses on the trend of data After an impact (Thiault et al., 2017). BACIPS approaches assume no trend in the Before period (Wauchope et al., 2021), and so this design assumes that the value of m_{eff} calculated before park designation had remained constant over time in both the Park and Control area.

Simple BACI designs can suffer from low statistical power, particularly if the number of replicates in the design is small (Christie et al., 2019; Thiault et al., 2019). Therefore, this study has assessed the entire National Parks System as a network as well as sub-networks based on their management strategy and geographic region, and also for each individual park-control pair, to estimate local effects of protection and to estimate effects of different management strategies across groups of protected areas (Underwood 1992; Osenberg et al., 2006; Rassweiler et al., 2021). Assessing the whole NPS, and sub-networks within it, has a higher statistical power than just assessing individual pairs (Thiault et al., 2019), and addresses the impact of the establishment of the individual protected areas, along with the changing goals of Parks Canada over its history and whether a park's region and remoteness from human activities affects its fragmentation levels.

3.2.5.2 Why use a Progressive-Change BACIPS approach?

The classic BACIPS approach assumes that an impact on an environment causes a step-change in the response variable, but it does not assess any trends in the response variable after the impact (Osenberg et al., 2006; Thiault et al., 2017; Kerr et al., 2019). Thiault et al. (2017) argue that just focusing on average change between the Before and After an impact can result in misleading estimates of effect size, due to the assumption that the magnitude of change remains constant. Therefore, their Progressive-Change BACIPS approach expands the scope of the original BACIPS by comparing multiple models and selecting the best-fit model with the underlying dynamics of the environmental system (Thiault et al., 2019; Wauchope et al., 2021). A step-change model is selected when the park designation has caused an immediate, or very quick change in the difference of effective mesh size between the impact and control areas. A linear model is selected when the park designation causes a continuous change in the difference at a constant rate. An asymptotic model is selected when the park designation causes a continuous change in the difference, but with the difference changing at a declining rate over time and approaching an asymptote. A sigmoid model is selected when the park designation causes a continuous change in the difference, but after the impact this shifts from initially accelerating to decelerating (Thiault et al., 2017). A Progressive-Change BACIPS approach is more flexible than traditional BACI or BACIPS designs, as the data are used to inform the final model. Varying trends through time are considered, accounting for any delayed or slow long-term changes. Despite the risk of sparse data sometimes leading to the selection of simpler (incorrect) models over more complex (true) models (Christie et al., 2019), these simple models may still provide a coarser but more reliable prediction of the effect of the protected area for pairs that have fewer time-steps available for analysis, i.e., for recently designated parks (Thiault et al., 2017).

Before progressing with the Progressive-Change BACIPS approach, the data was tested for the assumptions of normality, homoscedasticity, and the absence of autocorrelation.
The Progressive-Change BACIPS approach involves 4 main steps (Thiault et al., 2017):

1. Obtain the values of Effective Mesh size at the Park and Control sites Before and several dates After the intervention (designation of park status).
2. Derive the time series of differences (Δ) between the Park and Control sites.
3. Compare step-change, linear, asymptotic, and sigmoid models, and selected the best-fit model using AICc weighting.
4. Derive inferences from the selected model for the park-control pair or networks under consideration, which provides an estimate of the effect of park designation on landscape fragmentation.

The step-change model uses a simple Anova; the linear model uses a simple linear regression; and the asymptotic and sigmoid models use nonlinear regressions.

All analysis was completed in R version 4.2.2 (R Core Team, 2022), and used code from Thiault et al. (2017). R packages used were minpack.lm (Elzhov et al., 2023) and nsl2 (Grothendieck, 2024) for nonlinear regressions, and AICcmodavg (Mazerolle, 2023) for evaluating second-order AIC. The R code is given in the appendices.

3.3 Results

To ascertain whether park protection has prevented an increase in fragmentation over time compared to similar unprotected areas, this analysis covered each park-control pair, the National Parks System as a whole, regional groups of parks, and groups of parks based on when they were protected. The null hypothesis was that the fragmentation levels in the parks and control areas changed the same amount over time.

For each park-control pair, and each group and sub-group of pairs, the Progressive-Change BACIPS method fits the data shape to the most appropriate model using AICc weighting. This method gives results as the divergence of effective mesh size between the park and control areas from before park designation to after, as a magnitude of change (km^2), or the rate of change (km^2/yr). For the step-change, sigmoid and asymptotic models, the magnitude of the effect is defined as the difference before the intervention (i.e. park protection) and the eventual asymptote. For the linear model, the magnitude of the effect is defined as the rate at which the fragmentation levels in the park and control areas diverged from each other (Thiault et al., 2017).

3.3.1 Standalone park-control pairs

The main findings from the PC-BACIPS analysis for each park-control pair include the magnitude of the effect (the impact of park protection on fragmentation levels), its direction, and its significance. In the pair analysis, the most common model selected by AICc weighting was the linear model. Park-control pairs with a notable change at one time-step but that were stable at other points in time had the step-change model selected. For example, for FG1 in Elk Island and its control, many roads were paved between the 1950s and 1980s but otherwise FG1 elements remained stable. In addition, park-control pairs with fewer time-steps post-protection date, such as Tuktut Nogait, had the step-change model selected. Some park-control pairs where the date of protection fell between 1950 and 1980 had four data points, which forced the PC-BACIPS method to select the asymptotic or sigmoid models. Due to there being too few datapoints at the park-control pair level, a simple linear model or step model was selected instead, to avoid violating the assumptions of the asymptotic or sigmoid models (Christie et al., 2019).

35% of park-control pairs have a negative direction of effect, suggesting that at least locally, park protection has not prevented an increase in fragmentation over time compared to each control area. The divergence of effective mesh size indicates that the parks have had an increase in fragmentation levels more over time than in their associated control area. The most significant of these is in Banff, where there is an effect size of over 1 km^2/yr with high significance levels for FG1, 2 and 3, as demonstrated in table 3.3.1, and figure 3.3.1f. Other examples where park protection has not prevented increasing fragmentation are Mount Revelstoke (figure 3.3.1c), Terra Nova (figure 3.3.1d), Glacier, Yoho, Jasper, Forillon, La Mauricie, Nahanni, AkMM, Bruce Peninsula and Torngat Mountains. In the Wood Buffalo, Prince Albert, Kejimkujik, Kouchibouguac and Kluane park-control-pairs, the parks prevented the fragmentation associated with FG1, however the parks had a more significant decrease in effective mesh size for FG2 (table 3.3.1). In Kouchibouguac and La Mauricie (figure 3.3.1h), the differences between the CUT and CBC results suggest that park protection has had little effect in curbing fragmentation in the area adjacent to the park boundaries – the buffer zone.

In contrast, 15% of park-control pairs have had a positive effect direction, where park protection has successfully prevented fragmentation that would have occurred had the areas not been designated as national parks. This is especially evident in the Riding Mountain (figure 3.3.1g) and Elk Island park-control pairs, with their positive direction of effect and high effect size as shown in table 3.3.1. At Elk Island, the differences between the results for the CUT method and the CBC method for FG1 (CUT: 1.83 km^2; CBC: 11.19 km^2) and FG2 (CUT: 0.15 km^2/yr; CBC: 1.12 km^2/yr) indicate that the park protection has also prevented an increase in fragmentation just outside the boundaries of the national park (figure 3.3.1b)..

The Auyuittuq, Qausuittuq, Sirmilik, and Ukkusiksalik park-control pairs exhibited zero change in effective mesh size between before park designation and after in either the park or the control. This is due to their remoteness from human activity and the relatively short amount of time since park designation to present day- between 48 years for Auyuittuq and 17 years for Ukkusiksalik. Around 50% of park-control pairs exhibited minimal BACI change – indicating a neutral effect. An example of this is the Quttinirpaaq park-control pair, with an effect size of less than 0.2km^2, as shown in figure 3.3.1e.

Table 3.3.1 *- Results of the PC-BACIPS analysis for each park-control pair with effect estimate, direction, significance and likelihood of model type selection. The darkest green shading represents significant results (p<0.05), light green represents marginally significant results (0.1>p>0.05) and no shading indicates insignificant results. Models with no AICc weighting noted had their model chosen at 100% certainty. This occurred for park-control pairs with few time-steps measured.*

Park	**FG1**	**FG2**	**FG3**	**FG4**	**FG5**
Banff	CUT: **-1.17 km²/yr** *p*-value: 0.0006 CBC: **-1.66 km²/yr** *p*-value: 0.0007 Linear model CUT: 99.9%; CBC: 99.9%	CUT: **- 1 km²/yr** *p*-value: 0.001 CBC: **-1.46 km²/yr** *p*-value: 0.005 Linear model: 99.9%, 99.5%	CUT: **-0.909 km²/yr** *p*-value: 0.001 CBC: **-1.31 km²/yr** *p*-value: 0.005 Linear model: 99.9%, 99.5%	CUT: **-0.765 km²/yr** *p*-value: 0.0003 CBC: **-0.629 km²/yr** *p*-value: 0.03 Linear model: 99.9%, 97.1%	CUT: **-0.471 km²/yr** *p*-value: 0.004 CBC: **-1.28 km²/yr** *p*-value: 0.08 Linear model: 99.6%, 89.8%
Glacier	CUT: **-0.0498 km²/yr** *p*-value: 0.03 CBC: **-0.0841 km²/yr** *p*-value: 0.06 Linear model: 91.6%, 88%	CUT: **-0.0905 km²/yr** *p*-value: 0.003 CBC: **-0.187 km²/yr** *p*-value: 0.01 Linear model: 99.6%, 98.9%	CUT: **-0.0789 km²/yr** *p*-value: 0.003 CBC: **-0.179 km²/yr** *p*-value: 0.009 Linear model: 99.7%, 99%	CUT: **-0.0698 km²/yr** *p*-value: 0.003 CBC: **-0.147 km²/yr** *p*-value: 0.01 Linear model: 99.7%, 98.7%	CUT: **-0.0701 km²/yr** *p*-value: 0.003 CBC: **-0.148 km²/yr** *p*-value: 0.01 Linear model: 99.7%, 98.7%
Yoho	CUT: **-0.123 km²/yr** *p*-value: 0.001 CBC: **-0.3889 km²/yr** *p*-value: 0.0001 Linear model: 99.6%, 100%	CUT: **-0.116 km²/yr** *p*-value: 0.002 CBC: **-0.398 km²/yr** *p*-value: 0.0005 Linear model: 97.9%, 99.9%	CUT: **- 0.104 km²/yr** *p*-value: 0.001 CBC: **-0.358 km²/yr** *p*-value: 0.0003 Linear model: 98.7%, 99.9%	CUT: **-0.0806 km²/yr** *p*-value: 0.002 CBC: **-0.279 km²/yr** *p*-value: 0.0004 Linear model: 98.5%, 99.9%	CUT: **-0.08123 km²/yr** *p*-value: 0.002 CBC: **-0.282 km²/yr** *p*-value: 0.0004 Linear model: 98.4%, 99.9%
Waterton Lakes	CUT: **-0.194 km²/yr** *p*-value: 0.003 CBC: **-0.428 km²/yr** *p*-value: 0.003 Linear model: 97.4%, 96.3%	CUT: **-0.837 km²/yr** *p*-value: 0.2 CBC: **-2.989 km²/yr** *p*-value: 0.2 Linear model: 74.6%, 76%	CUT: **-0.752 km²/yr** *p*-value: 0.2 CBC: **-2.71 km²/yr** *p*-value: 0.2 Linear model: 73.9%, 75.6%	CUT: **-0.752 km²/yr** *p*-value: 0.2 CBC: **-2.71 km²/yr** *p*-value: 0.2 Linear model: 73.9%, 75.5%	CUT: **-0.778 km²/yr** *p*-value: 0.2 CBC: **-2.8 km²/yr** *p*-value: 0.2 Linear model: 73.9%, 75.6%
Jasper	CUT: **-0.303 km²/yr** *p*-value: 0.03 CBC: **-0.452 km²/yr** *p*-value: 0.02 Linear model: 51.9%, 64.1%	CUT: **-0.317 km²/yr** *p*-value: 0.02 CBC: **-0.478 km²/yr** *p*-value: 0.01 Linear model: 96.4%, 98.1%	CUT: **-0.281 km²/yr** *p*-value: 0.01 CBC: **-0.412 km²/yr** *p*-value: 0.008 Linear model: 96.8%, 98.3%	CUT: **-0.213 km²/yr** *p*-value: 0.02 CBC: **-0.316 km²/yr** *p*-value: 0.01 Linear model: 95.6%, 97.6%	CUT: **-0.216 km²/yr** *p*-value: 0.02 CBC: **-0.320 km²/yr** *p*-value: 0.01 Linear model: 95.6%, 97.6%
Elk Island	CUT: **1.83 km²** *p*-value: 0.1 CBC: **11.2 km²** *p*-value: 0.004 Step model: 86.2%, 98.2%	CUT: **0.152 km²/yr** *p*-value: 0.08 CBC: **1.12 km²/yr** *p*-value: 0.05 Linear model: 88.8%, 93.3%	CUT: **0.139 km²/yr** *p*-value: 0.07 CBC: **0.988 km²/yr** *p*-value: 0.05 Linear model: 90%, 93.5%	CUT: **0.154 km²/yr** *p*-value: 0.08 CBC: **1.11 km²/yr** *p*-value: 0.05 Linear model: 90%, 93.4%	CUT: **0.154 km²/yr** *p*-value: 0.08 CBC: **1.11 km²/yr** *p*-value: 0.05 Linear model: 90%, 93.4%
Mount Revelstoke	CUT: **-0.0222 km²/yr** *p*-value: 0.05 CBC: **-0.126 km²/yr** *p*-value: 0.008 Linear model: 78.9%, 96%	CUT: **1.88 km²** *p*-value: 0.02 CBC: **-0.279 km²/yr** *p*-value: 0.04 Step model: 92.8%, Linear model: 86.6%	CUT**: 1.77 km²** *p*-value: 0.02 CBC: **11 km²** *p*-value: 0.03 Step model: 92.7%, 50.7%	CUT: **1.7 km²** *p*-value: 0.02 CBC: **-0.142 km²/yr** *p*-value: 0.02 Step model: 93.7%, Linear model: 53.7%	CUT: **1.7 km²** *p*-value: 0.02 CBC: **-0.142 km²/yr** *p*-value: 0.02 Step model: 94.7%, Linear model: 53.7%
Point Pelee	CUT: **-0.0202 km²/yr**	CUT: **-0.031 km²/yr**	CUT: **-0.0218 km²/yr**	CUT: **-0.0218 km²/yr**	CUT: **-0.0267 km²/yr**

	p-value: 0.1 CBC: **-0.157 km²/yr** *p*-value: 0.02 Linear model: 55.5%, 71.4%	*p*-value: 0.02 CBC: **-0.0587 km²/yr** *p*-value: 0.4 Linear model: 92.8%, 54.1%	*p*-value: 0.04 CBC: **-0.0487 km²/yr** *p*-value: 0.3 Linear model: 85.7%, 54.5%	*p*-value: 0.04 CBC: **-0.0487 km²/yr** *p*-value: 0.3 Linear model: 85.7%, 54.5%	*p*-value: 0.04 CBC: **-0.0606 km²/yr** *p*-value: 0.3 Linear model: 84.9%, 54.7%
Kootenay	CUT: **-0.0181 km²/yr** *p*-value: 0.2 CBC: **-0.272 km²/yr** *p*-value: 0.003 Linear model: 64.3%, 99.5%	CUT: **-0.307 km²/yr** *p*-value: 0.1 CBC: **0.244 km²/yr** *p*-value: 0.4 Linear model: 83.9%, 57.6%	CUT: **-0.289 km²/yr** *p*-value: 0.1 CBC: **0.269 km²/yr** *p*-value: 0.4 Linear model: 82.9%, 59.6%	CUT: **-0.254 km²/yr** *p*-value: 0.1 CBC: **0.321 km²/yr** *p*-value: 0.2 Linear model: 82%, 64.5%	CUT: **-0.256 km²/yr** *p*-value: 0.1 CBC: **0.323 km²/yr** *p*-value: 0.2 Linear model: 82%, 64.4%
Wood Buffalo	CUT: **0.49 km²/yr** *p*-value: 0.0009 CBC: **0.631 km²/yr** *p*-value: 0.001 Linear model: 99.9, 99.8%	CUT: **-3.12 km²/yr** *p*-value: 0.08 CBC: **-3.51 km²/yr** *p*-value: 0.09 Linear model: 87.2%, 86.2%	CUT: **-1.07 km²/yr** *p*-value: 0.05 CBC: **-3.27 km²/yr** *p*-value: 0.09 Linear model: 91.7%, 86.1%	CUT: **-1.07 km²/yr** *p*-value: 0.05 CBC: **-3.27 km²/yr** *p*-value: 0.09 Linear model: 91.7%, 86.1%	CUT: **-0.1 km²/yr** *p*-value: 0.07 CBC: **-3.43 km²/yr** *p*-value: 0.09 Linear model: 88.9%, 86.2%
Prince Albert	CUT: **0.0832 km²/yr** *p*-value: 0.3 CBC: **0.0666 km²/yr** *p*-value: 0.6 Linear model: 72.1%, 55.7%	CUT: **-0.874 km²/yr** *p*-value: 0.2 CBC: **-0.816 km²/yr** *p*-value: 0.3 Linear model: 80.4%, 67.6%	CUT: **-0.602 km²/yr** *p*-value: 0.2 CBC: **-0.506 km²/yr** *p*-value: 0.4 Linear model: 76.5%, 61.9%	CUT: **-0.602 km²/yr** *p*-value: 0.2 CBC: **-0.506 km²/yr** *p*-value: 0.4 Linear model: 76.5%, 61.9%	CUT: **-0.67 km²/yr** *p*-value: 0.2 CBC: **-0.548 km²/yr** *p*-value: 0.4 Linear model: 76.4%, 61.4%
Riding Mountain	CUT**: 0.16 km²/yr** *p*-value: 0.03 CBC: **0.273 km²/yr** *p*-value: 0.02 Linear model: 95.5%, 96.6%	CUT: **3.1 km²/yr** *p*-value: 0.01 CBC: **3.25 km²/yr** *p*-value: 0.02 Linear model: 98.2%, 98%	CUT: **3.12 km²/yr** *p*-value: 0.01 CBC: **2.8 km²/yr** *p*-value: 0.02 Linear model: 98.3%, 97.9%	CUT: **3.12 km²/yr** *p*-value: 0.01 CBC: **2.8 km²/yr** *p*-value: 0.02 Linear model: 98.3%, 97.9%	CUT**: 2.91 km²/yr** *p*-value: 0.01 CBC: **2.95 km²/yr** *p*-value: 0.02 Linear model: 98.2%, 98.9%
Terra Nova	CUT: **-0.057 km²/yr** *p*-value: 0.02 CBC: **-0.106 km²/yr** *p*-value: 0.02 Linear model	CUT: **-0.117 km²/yr** *p*-value: 0.004 CBC: **-0.215 km²/yr** *p*-value: 0.002 Linear model	CUT: **-0.107 km²/yr** *p*-value: 0.003 CBC: **-0.199 km²/yr** *p*-value: 0.0007 Linear model	CUT: **-0.107 km²/yr** *p*-value: 0.003 CBC: **-0.199 km²/yr** *p*-value: 0.0007 Linear model	CUT**: -0.115 km²/yr** *p*-value: 0.003 CBC: **-0.214 km²/yr** *p*-value: 0.001 Linear model
Kejimkujik	CUT: **0.00515 km²/yr** *p*-value: 0.9 CBC: **0.109 km²/yr** *p*-value: 0.5 Linear model	CUT: **-0.0352 km²/yr** *p*-value: 0.3 CBC: **-0.212 km²/yr** *p*-value: 0.3 Linear model	CUT: **-0.0273 km²/yr** *p*-value: 0.4 CBC: **-0.18 km²/yr** *p*-value: 0.3 Linear model	CUT: **-0.0273 km²/yr** *p*-value: 0.4 CBC: **-0.18 km²/yr** *p*-value: 0.3 Linear model	CUT: **-0.0323 km²/yr** *p*-value: 0.4 CBC: **-0.211 km²/yr** *p*-value: 0.3 Linear model
Kouchibouguac	CUT: **0.00102 km²/yr** *p*-value: 0.9 CBC: **0.0355 km²/yr** *p*-value: 0.8 Sigmoid model	CUT: **-0.375 km²/yr** *p*-value: 0.009 CBC: **-1.03 km²/yr** *p*-value: 0.04 Sigmoid model	CUT: **-0.289 km²/yr** *p*-value: 0.008 CBC: **-0.66 km²/yr** *p*-value: 0.05 Sigmoid model	CUT: **-0.289 km²/yr** *p*-value: 0.008 CBC: **-0.66 km²/yr** *p*-value: 0.05 Sigmoid model	CUT: **-0.343 km²/yr** *p*-value: 0.008 CBC: **-0.775 km²/yr** *p*-value: 0.05 Sigmoid model
Forillon	CUT: **-0.0668 km²/yr** *p*-value: 0.05 CBC: **-0.194 km²/yr** *p*-value: 0.03 Sigmoid model	CUT: **-0.0305 km²/yr** *p*-value: 0.5 CBC: **-0.0294 km²/yr** *p*-value: 0.6 Sigmoid model	CUT: **-0.0236 km²/yr** *p*-value: 0.5 CBC: **-0.0172 km²/yr** *p*-value: 0.7 Sigmoid model	CUT: **-0.0236 km²/yr** *p*-value: 0.5 CBC: **-0.0172 km²/yr** *p*-value: 0.7 Sigmoid model	CUT: **-0.0239 km²/yr** *p*-value: 0.5 CBC: **-0.0172 km²/yr** *p*-value: 0.7 Sigmoid model
La Mauricie	CUT: **-0.076 km²/yr** *p*-value: 0.5 CBC: **-0.0217 km²/yr** *p*-value: 0.9	CUT: **-0.68 km²/yr** *p*-value: 0.004 CBC: **-1.78 km²/yr** *p*-value: 0.008	CUT: **-0.615 km²/yr** *p*-value: 0.005 CBC: **-1.61 km²/yr** *p*-value: 0.009	CUT: **-0.615 km²/yr** *p*-value: 0.005 CBC: **-1.61 km²/yr** *p*-value: 0.009	CUT: **-0.648 km²/yr** *p*-value: 0.005 CBC: **-1.69 km²/yr** *p*-value: 0.009

	Linear model	Linear model	Linear model	Linear model	Linear model
Pacific Rim	CUT: **-0.0282 km^2/yr** *p*-value: 0.1 CBC: **0.289 km^2/yr** *p*-value: 0.09 Sigmoid model	CUT: **-0.0437 km^2/yr** *p*-value: 0.02 CBC: **0.168 km^2/yr** *p*-value: 0.1 Sigmoid model	CUT: **-0.0104 km^2/yr** *p*-value: 0.2 CBC: **0.217 km^2/yr** *p*-value: 0.09 Sigmoid model	CUT: **-0.0104 km^2/yr** *p*-value: 0.2 CBC: **0.217 km^2/yr** *p*-value: 0.09 Sigmoid model	CUT: **-0.0309 km^2/yr** *p*-value: 0.06 CBC: **0.151 km^2/yr** *p*-value: 0.1 Sigmoid model
Auyuittuq	CUT: **0** *p*-value: 0 CBC: **0.000153 km^2/yr** *p*-value: 0.01 Sigmoid model	CUT: **0** *p*-value: 0 CBC: **0.000153 km^2/yr** *p*-value: 0.01 Sigmoid model	CUT: **0** *p*-value: 0 CBC: **-0.01 km^2/yr** *p*-value: 0.01 Sigmoid model	CUT: **0** *p*-value: 0 CBC: **-0.0108 km^2/yr** *p*-value: 0.01 Sigmoid model	CUT: **0** *p*-value: 0 CBC: **-0.0116 km^2/yr** *p*-value: 0.01 Sigmoid model
Kluane	CUT: **0.0183 km^2/yr** *p*-value: 0.8 CBC: **0.0133 km^2/yr** *p*-value: 0.8 Sigmoid model	CUT: **-0.0728 km^2/yr** *p*-value: 0.5 CBC: **-0.117 km^2/yr** *p*-value: 0.3 Sigmoid model	CUT: **-0.0519 km^2/yr** *p*-value: 0.6 CBC: **-0.0764 km^2/yr** *p*-value: 0.5 Sigmoid model	CUT: **-0.0511 km^2/yr** *p*-value: 0.5 CBC: **-0.0706 km^2/yr** *p*-value: 0.5 Sigmoid model	CUT: **-0.0517 km^2/yr** *p*-value: 0.5 CBC: **-0.0718 km^2/yr** *p*-value: 0.5 Sigmoid model
Nahanni	CUT: **-0.00119 km^2/yr** *p*-value: 0.9 CBC: **-0.00932 km^2/yr** *p*-value: 0.7 Linear model	CUT: **-0.00119 km^2/yr** *p*-value: 0.9 CBC: **0.0248 km^2/yr** *p*-value: 0.2 Linear model	CUT: **-0.00115 km^2/yr** *p*-value: 0.9 CBC: **-0.00599 km^2/yr** *p*-value: 0.1 Linear model	CUT: **-0.000859 km^2/yr** *p*-value: 0.9 CBC: **-0.0064 km^2/yr** *p*-value: 0.4 Linear model	CUT: **-0.000768 km^2/yr** *p*-value: 0.9 CBC: **-0.00639 km^2/yr** *p*-value: 0.4 Linear model
Gros Morne	CUT: **-0.0966 km^2/yr** *p*-value: 0.04 CBC: **-0.176 km^2/yr** *p*-value: 0.02 Sigmoid model	CUT: **-0.0259 km^2/yr** *p*-value: 0.7 CBC: **-0.122 km^2/yr** *p*-value: 0.2 Sigmoid model	CUT: **-0.0245 km^2/yr** *p*-value: 0.6 CBC: **-0.107 km^2/yr** *p*-value: 0.2 Sigmoid model	CUT: **-0.0245 km^2/yr** *p*-value: 0.6 CBC: **-0.107 km^2/yr** *p*-value: 0.2 Sigmoid model	CUT: **-0.0154 km^2/yr** *p*-value: 0.8 CBC: **-0.0988 km^2/yr** *p*-value: 0.3 Sigmoid model
Pukaskwa	CUT: **-0.00559 km^2/yr** *p*-value: 0.01 CBC: **-0.0118 km^2/yr** *p*-value: 0.03 Sigmoid model	CUT: **0.0711 km^2/yr** *p*-value: 0.006 CBC: **0.128 km^2/yr** *p*-value: 0.007 Sigmoid model	CUT: **0.065 km^2/yr** *p*-value: 0.006 CBC: **0.116 km^2/yr** *p*-value: 0.007 Sigmoid model	CUT: **0.065 km^2/yr** *p*-value: 0.006 CBC: **0.116 km^2/yr** *p*-value: 0.007 Sigmoid model	CUT: **0.0697 km^2/yr** *p*-value: 0.006 CBC: **0.125 km^2/yr** *p*-value: 0.007 Sigmoid model
Grasslands	CUT: **0.000587 km^2/yr** *p*-value: 0.2 CBC: **0.0101 km^2/yr** *p*-value: 0.02 Asymptotic model	CUT: **0.872 km^2/yr** *p*-value: 0.1 CBC: **0.597 km^2/yr** *p*-value: 0.8 Sigmoid model	CUT: **0.83 km^2/yr** *p*-value: 0.1 CBC: **0.569 km^2/yr** *p*-value: 0.8 Sigmoid model	CUT: **0.83 km^2/yr** *p*-value: 0.1 CBC: **0.569 km^2/yr** *p*-value: 0.8 Sigmoid model	CUT: **0.84 km^2/yr** *p*-value: 0.09 CBC: **0.611 km^2/yr** *p*-value: 0.8 Sigmoid model
Ivvavik	CUT: **-0.000938 km^2/yr** *p*-value: 0.2 CBC: **-0.00109 km^2/yr** *p*-value: 0.2 Sigmoid model	CUT: **-0.023 km^2/yr** *p*-value: 0.7 CBC: **-0.0304 km^2/yr** *p*-value: 0.6 Sigmoid model	CUT: **-0.0223 km^2/yr** *p*-value: 0.6 CBC: **-0.0295 km^2/yr** *p*-value: 0.6 Sigmoid model	CUT: **-0.0221 km^2/yr** *p*-value: 0.6 CBC: **-0.0291 km^2/yr** *p*-value: 0.6 Sigmoid model	CUT: **-0.0225 km^2/yr** *p*-value: 0.6 CBC: **-0.0296 km^2/yr** *p*-value: 0.6 Sigmoid model
Bruce Peninsula	CUT: **-0.0380 km^2/yr** *p*-value: 0.2 CBC: **-0.138 km^2/yr** *p*-value: 0.3 Sigmoid model	CUT: **-0.156 km^2/yr** *p*-value: 0.3 CBC: **-0.613 km^2/yr** *p*-value: 0.1 Sigmoid model	CUT: **-0.151 km^2/yr** *p*-value: 0.3 CBC: **-0.562 km^2/yr** *p*-value: 0.09 Sigmoid model	CUT: **-0.151 km^2/yr** *p*-value: 0.3 CBC: **-0.562 km^2/yr** *p*-value: 0.09 Sigmoid model	CUT: **-0.16 km^2/yr** *p*-value: 0.3 CBC: **-0.594 km^2/yr** *p*-value: 0.09 Sigmoid model
Gwaii Haanas	CUT: **0.0107 km^2/yr** *p*-value: 0.3	CUT: **-0.148 km^2/yr** *p*-value: 0.3	CUT: **-0.0198 km^2/yr** *p*-value: 0.5	CUT: **-0.0198 km^2/yr** *p*-value: 0.5	CUT: **-1.67 km^2/yr** *p*-value: 0.2

	CBC: **0.00504 km^2/yr** *p*-value: 0.4 Sigmoid model	CBC: **-0.148 km^2/yr** *p*-value: 0.3 Asymptotic model	CBC: **-0.136 km^2/yr** *p*-value: 0.3 Asymptotic model	CBC: **-0.136 km^2/yr** *p*-value: 0.3 Asymptotic model	CBC: **-0.139 km^2/yr** *p*-value: 0.3 Asymptotic model
Quttinirpaaq	CUT: **0.17 km^2** *p*-value: 2.00E-16 CBC: **0.185 km^2** *p*-value: 2.00E-16 Step model	CUT: **0.17 km^2** *p*-value: 2.00E-16 CBC: **0.185 km^2** *p*-value: 2.00E-16 Step model	CUT: **0.0788 km^2** *p*-value: 2.00E-16 CBC: **0.0865 km^2** *p*-value: 2.00E-16 Step model	CUT: **0.093 km^2** *p*-value: 2.00E-16 CBC: **0.0718 km^2** *p*-value: 2.00E-16 Step model	CUT: **0.102 km^2** *p*-value: 2.00E-16 CBC: **0.0788 km^2** *p*-value: 2.00E-16 Step model
Aulavik	CUT: **-0.0199 km^2** *p*-value: 2.00E-16 CBC: **-0.0238 km^2** *p*-value: 2.00E-16 Step model	CUT: **-0.2 km^2** *p*-value: 6.00E-10 CBC: **-0.261 km^2** *p*-value: 4.00E-10 Step model	CUT: **-0.117 km^2** *p*-value: 2.00E-09 CBC: **-0.161 km^2** *p*-value: 1.00E-09 Step model	CUT: **-0.117 km^2** *p*-value: 2.00E-09 CBC: **-0.161 km^2** *p*-value: 1.00E-09 Step model	CUT: **0.895 km^2** *p*-value: 3.00E-11 CBC: **0.487 km^2** *p*-value: 2.00E-16 Step model
Vuntut	CUT: **0.0277 km^2/yr** *p*-value: 0.01 CBC: **0.0398 km^2/yr** *p*-value: 0.008 Linear model	CUT: **0.237 km^2/yr** *p*-value: 0.02 CBC: **0.356 km^2/yr** *p*-value: 0.02 Linear model	CUT: **0.0287 km^2/yr** *p*-value: 0.5 CBC: **0.305 km^2/yr** *p*-value: 0.02 Linear model	CUT: **0.0287 km^2/yr** *p*-value: 0.5 CBC: **0.305 km^2/yr** *p*-value: 0.02 Linear model	CUT: **0.0883 km^2/yr** *p*-value: 0.02 CBC: **0.381 km^2/yr** *p*-value: 0.02 Linear model
Wapusk	CUT: **0.000234 km^2/yr** *p*-value: 0.2 CBC: **0.000313 km^2/yr** *p*-value: 0.4 Linear model	CUT: **0.000234 km^2/yr** *p*-value: 0.2 CBC: **-0.0951 km^2/yr** *p*-value: 0.3 Linear model	CUT: **0.000773 km^2/yr** *p*-value: 0.3 CBC: **-0.0923 km^2/yr** *p*-value: 0.3 Linear model	CUT: **0.000773 km^2/yr** *p*-value: 0.3 CBC: **-0.0923 km^2/yr** *p*-value: 0.3 Linear model	CUT: **0.0132 km^2/yr** *p*-value: 0.4 CBC: **-0.0896 km^2/yr** *p*-value: 0.3 Linear model
Tuktut Nogait	CUT: **-0.345 km^2** *p*-value: 2.00E-16 CBC: **-0.414 km^2** *p*-value: 2.00E-16 Step model	CUT: **-0.345 km^2** *p*-value: 2.00E-16 CBC: **-0.357 km^2** *p*-value: 2.00E-16 Step model	CUT: **-0.310 km^2** *p*-value: 2.00E-16 CBC: **-0.321 km^2** *p*-value: 2.00E-16 Step model	CUT: **-0.31 km^2** *p*-value: 2.00E-16 CBC: **-0.321 km^2** *p*-value: 2.00E-16 Step model	CUT: **-0.341 km^2** *p*-value: 2.00E-16 CBC: **-0.348 km^2** *p*-value: 2.00E-16 Step model
Sirmilik	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC**: 0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na
Gulf Islands	CUT: **-0.00491 km^2/yr** *p*-value: 0.6 CBC: **-0.0124 km^2/yr** *p*-value: 0.6 Sigmoid model	CUT: **0.00993 km^2/yr** *p*-value: 0.4 CBC: **0.0511 km^2/yr** *p*-value: 0.1 Sigmoid model	CUT: **-0.0198 km^2/yr** *p*-value: 0.4 CBC: **-0.0274 km^2/yr** *p*-value: 0.6 Sigmoid model	CUT: **-0.0198 km^2/yr** *p*-value: 0.4 CBC: **-0.0274 km^2/yr** *p*-value: 0.6 Sigmoid model	CUT: **-0.0275 km^2/yr** *p*-value: 0.3 CBC: **-0.0503 km^2/yr** *p*-value: 0.4 Sigmoid model
Ukkusiksalik	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na
Torngat Mountains	CUT: **-0.000161 km^2** *p*-value: 7.00E-11 CBC: **-0.00016 km^2** *p*-value: 2.00E-10 Step model	CUT: **-0.00855 km^2** *p*-value: 3.00E-12 CBC: **-0.0105 km^2** *p*-value: 2.00E-13 Step model	CUT: **-0.00343 km^2** *p*-value: 3.00E-11 CBC: **-0.00892 km^2** *p*-value: 2.00E-11 Step model	CUT: **-0.237 km^2** *p*-value: 3.00E-14 CBC: **-0.784 km^2** *p*-value: 3.00E-13 Step model	CUT: **1.34 km^2** *p*-value: 2.00E-14 CBC: **2.22 km^2** *p*-value: 2.00E-14 Step model
Nááts'įhch'oh	CUT: **-0.0579 km^2** *p*-value: 5.00E-15	CUT: **-0.122 km^2** *p*-value: 2.00E-16	CUT: **0.111 km^2** *p*-value: 2.00E-12	CUT: **0.0542 km^2** *p*-value: 8.00E-12	CUT: **0.0553 km^2** *p*-value: 4.00E-12

	CBC: **-0.251 km²** *p*-value: 6.00E-16 Step model	CBC: **-0.767 km²** *p*-value: 1.00E-15 Step model	CBC: **-0.74 km²** *p*-value: 1.00E-14 Step model	CBC: **-0.00985 km²** *p*-value: 5.00E-11 Step model	CBC: **-0.00831 km²** *p*-value: 7.00E-11 Step model
Akami-uapishkU-KakKasuak-Mealy Mountains	CUT: **-1.43 km²** *p*-value: 2.00E-15 CBC: **-1.79 km²** *p*-value: 4.00E-14 Step model	CUT: **-1.42 km²** *p*-value: 4.00E-15 CBC: **-1.78 km²** *p*-value: 2.00E-13 Step model	CUT: **-1.31 km²** *p*-value: 9.00E-15 CBC: **-1.34 km²** *p*-value: 9.00E-14 Step model	CUT: **-1.31 km²** *p*-value: 9.00E-16 CBC: **-1.34 km²** *p*-value: 7.00E-14 Step model	CUT: **-32.2 km²** *p*-value: 2.00E-16 CBC: **-38.3 km²** *p*-value: 7.00E-15 Step model
Qausuittuq	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na	CUT: **0** *p*-value: 0 CBC: **0** *p*-value: 0 Model: na
Thaidene Nëné	CUT: **-0.000001 km²/yr** *p*-value: 3.00E-11 CBC**: 0.0122 km²/yr** *p*-value: 2.00E-14 Linear model	CUT: **-0.000001 km²/yr** *p*-value: 3.00E-11 CBC: **-0.0414 km²/yr** *p*-value: 6.00E-15 Linear model	CUT: **-0.034 km²/yr** *p*-value: 4.00E-11 CBC: **-0.162 km²/yr** *p*-value: 1.00E-12 Linear model	CUT: **-0.034 km²/yr** *p*-value: 4.00E-11 CBC: **-0.162 km²/yr** *p*-value: 1.00E-12 Linear model	CUT: **-13.9 km²/yr** *p*-value: 7.00E-14 CBC: **-4.54 km²/yr** *p*-value: 2.00E-14 Linear model

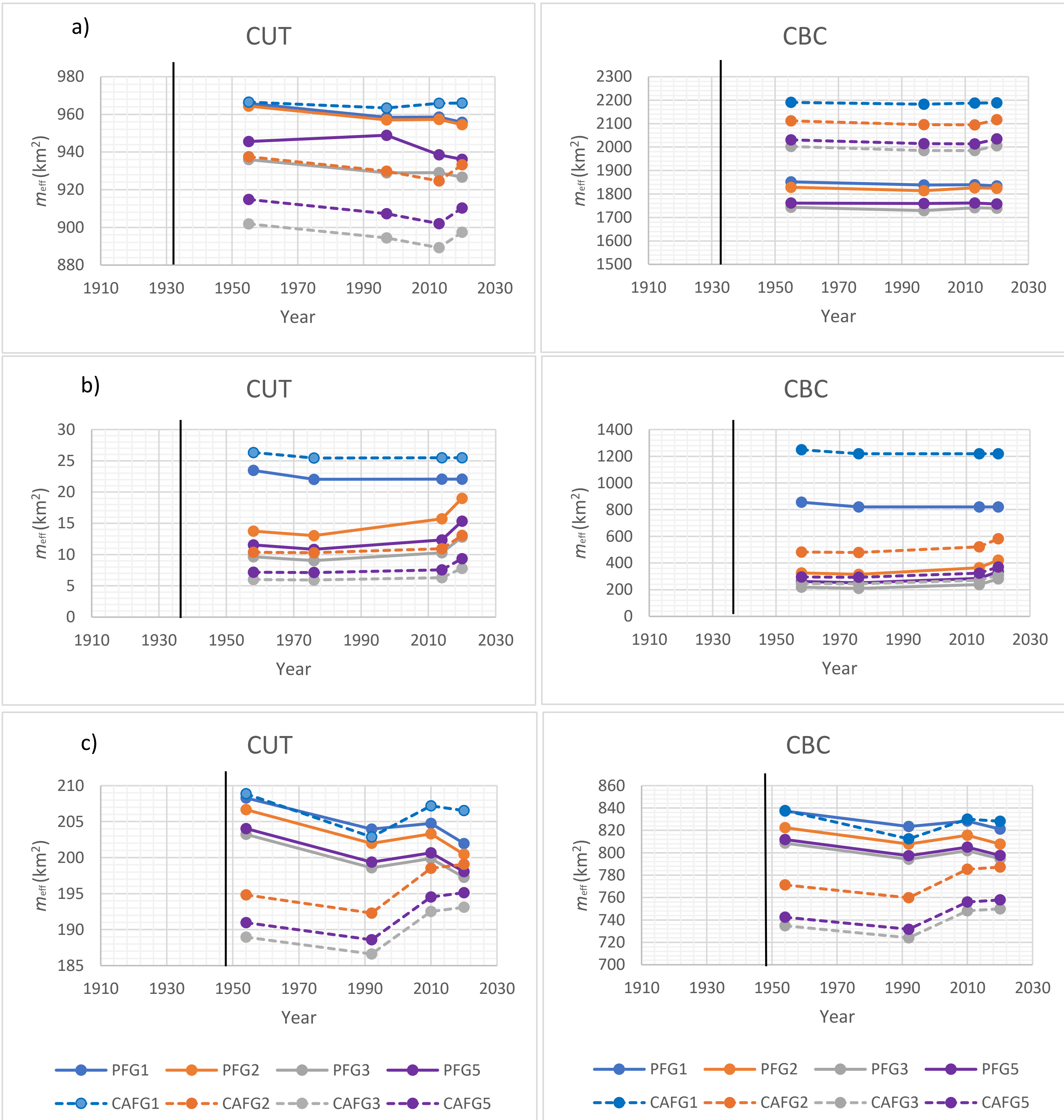


Figure 3.3.1a- *Changes over time in the a) Cape Breton Highlands, b) Prince Edward Island and c) Fundy park-control pairs. These parks were established in 1936, 1937, and 1948 respectively, indicated by the vertical lines. Owing to poor early topographic map data, these pairs could not be included in the statistical analysis, but still demonstrate change over time in fragmentation levels between the park and control area. PFG1-PFG5 = park values, CAFG1-CAFG5 = control area values.*

In the CBH, PEI, and Fundy park-control pairs, there were not adequate topographic map data for the earliest time-step and so these pairs are lacking any "Before" data. These were unable to be included in the statistical analysis, but their trends in effective mesh size are shown in figure 3.3.1a. The CBH pair demonstrates higher fragmentation in the park than the control area over time, across all FGs. In PEI, fragmentation levels are stable for FG1, however for FG2 onwards, both the park and control area showed an increase in effective mesh size, particularly after 2012. The Fundy park-control pair demonstrated an overall decrease in effective mesh size over time in the park, but an increase in the control area.

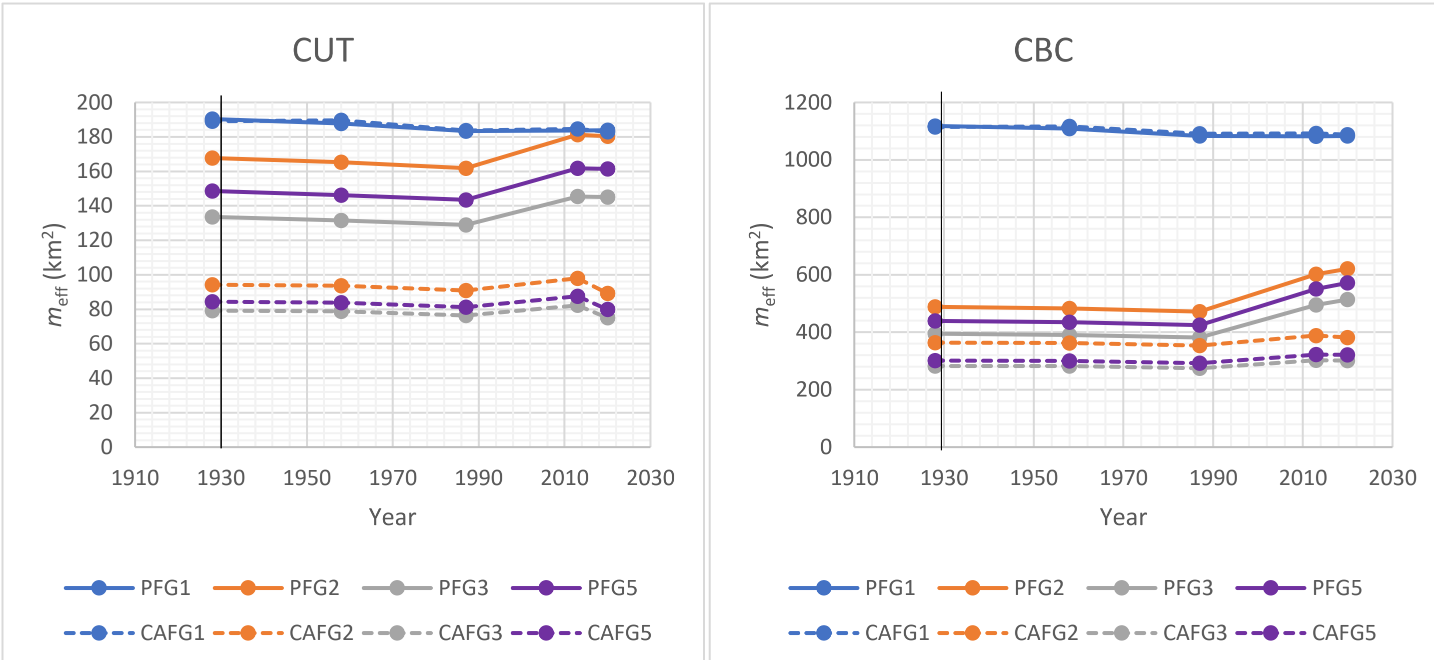


Figure 3.3.1b- *Changes over time in the Elk Island park-control pair, protected in 1913 and conserved with the National Parks Act of 1930. In almost all instances, except FG1 at the earliest time step pre-1930, Elk Island park has a higher effective mesh size than its paired control area, increasing over time. Elk Island is the only park in the NPS which is fully fenced in. Therefore, for most true-to-life results, it may be most useful to only consider the CUT values rather than the CBC for this park.*

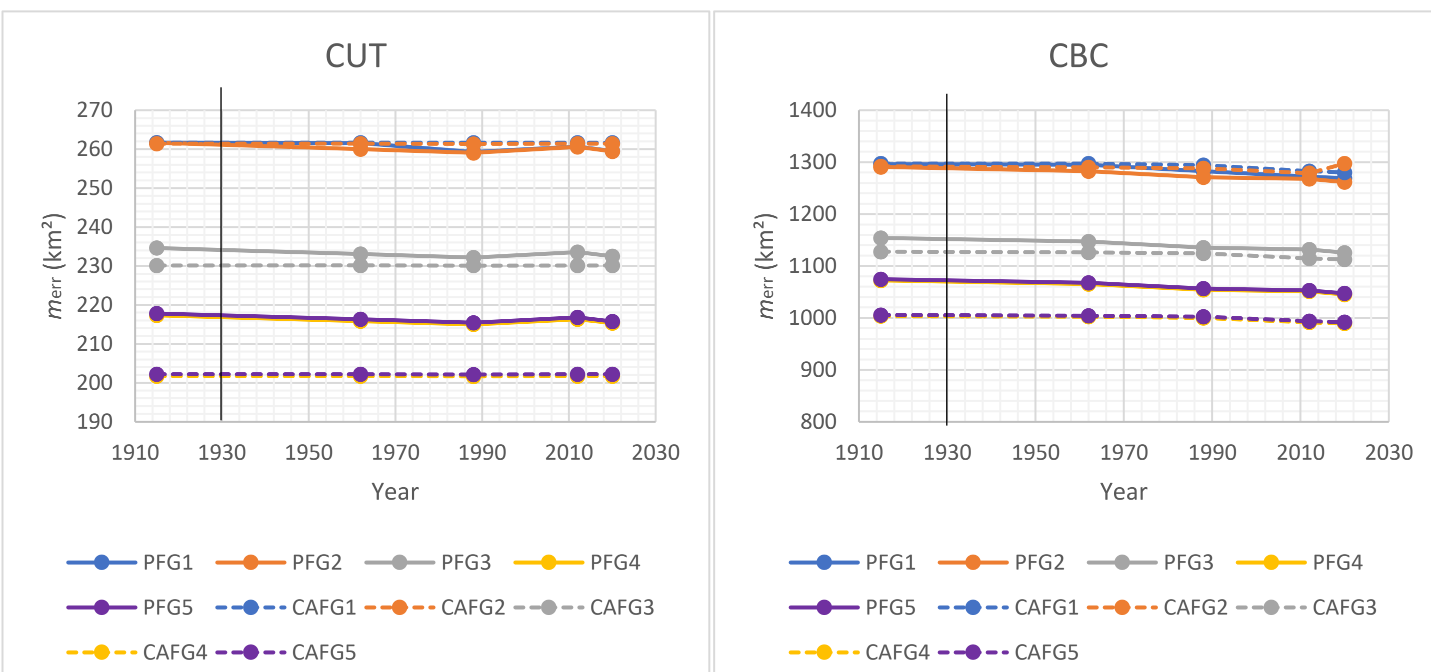


Figure 3.3.1c- *Changes over time in the Mount Revelstoke park-control pair, protected in 1914. Effective mesh size decreases over time in both the park and the control area for most FGs, with the control area having greater effective mesh size than the park for FG1 and FG2, and vice versa for the other FGs.*

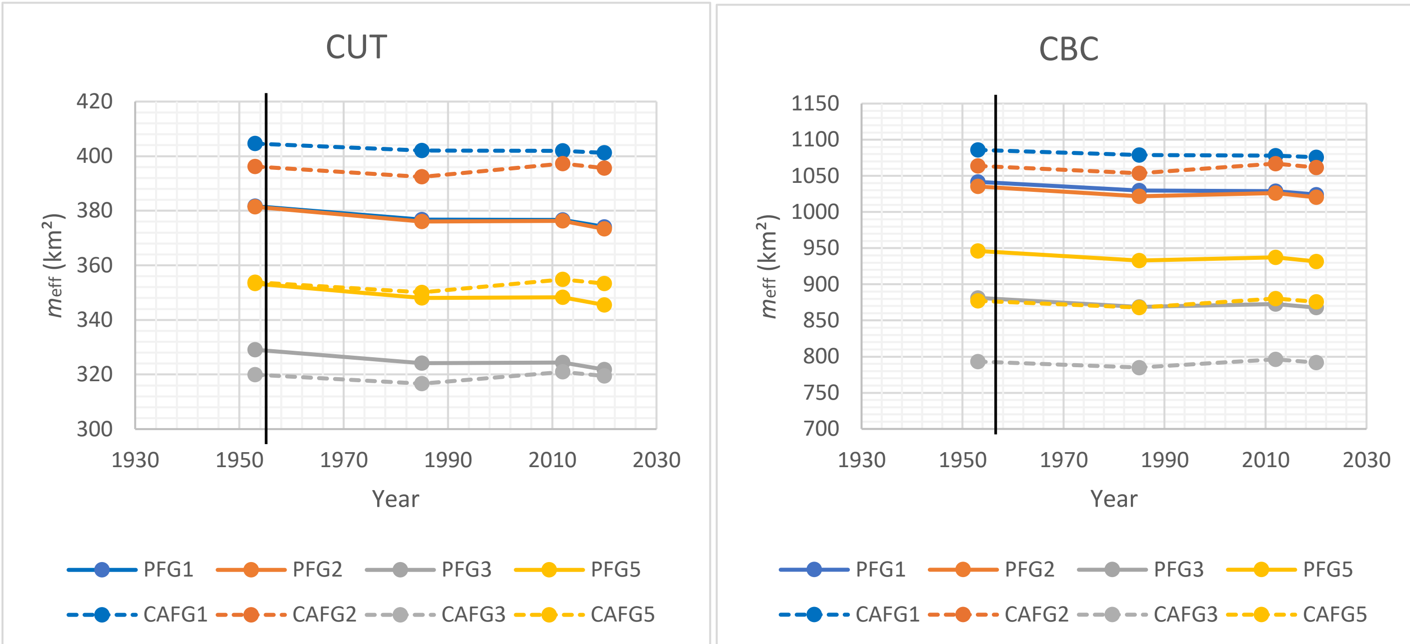


Figure 3.3.1d- *Changes over time in the Terra Nova park-control pair, protected in 1957. Effective mesh size decreases over time in both the park and the control area for all FGs, with the control area having greater effective mesh size than the park except in FG3 (CUT), and FG3 and FG5 (CBC), where natural fragmenting elements play a greater factor in the control area.*

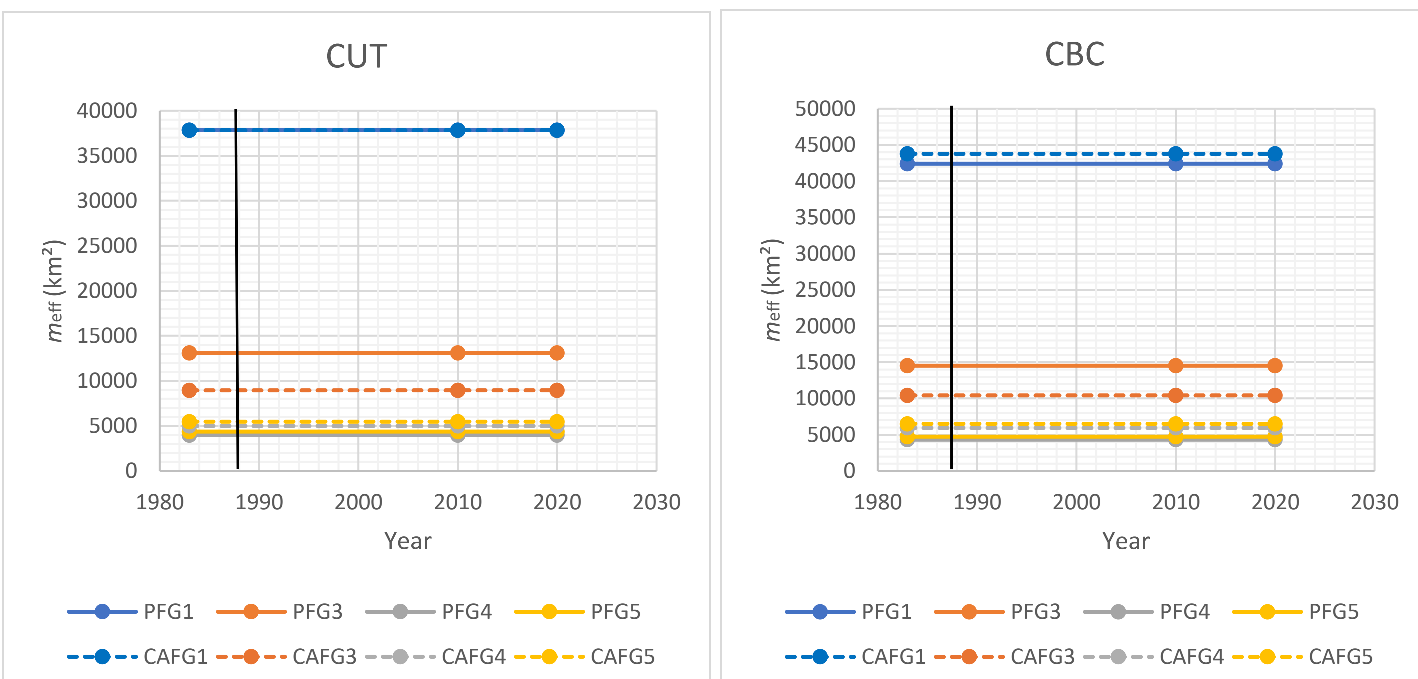


Figure 3.3.1e- *Changes over time in the Quttinirpaaq park-control pair, protected in 1988. Effective mesh size has remained stable over time with very minimal change. The park has a greater effective mesh size for FG2, but the control area has greater effective mesh size in all other FGs.*

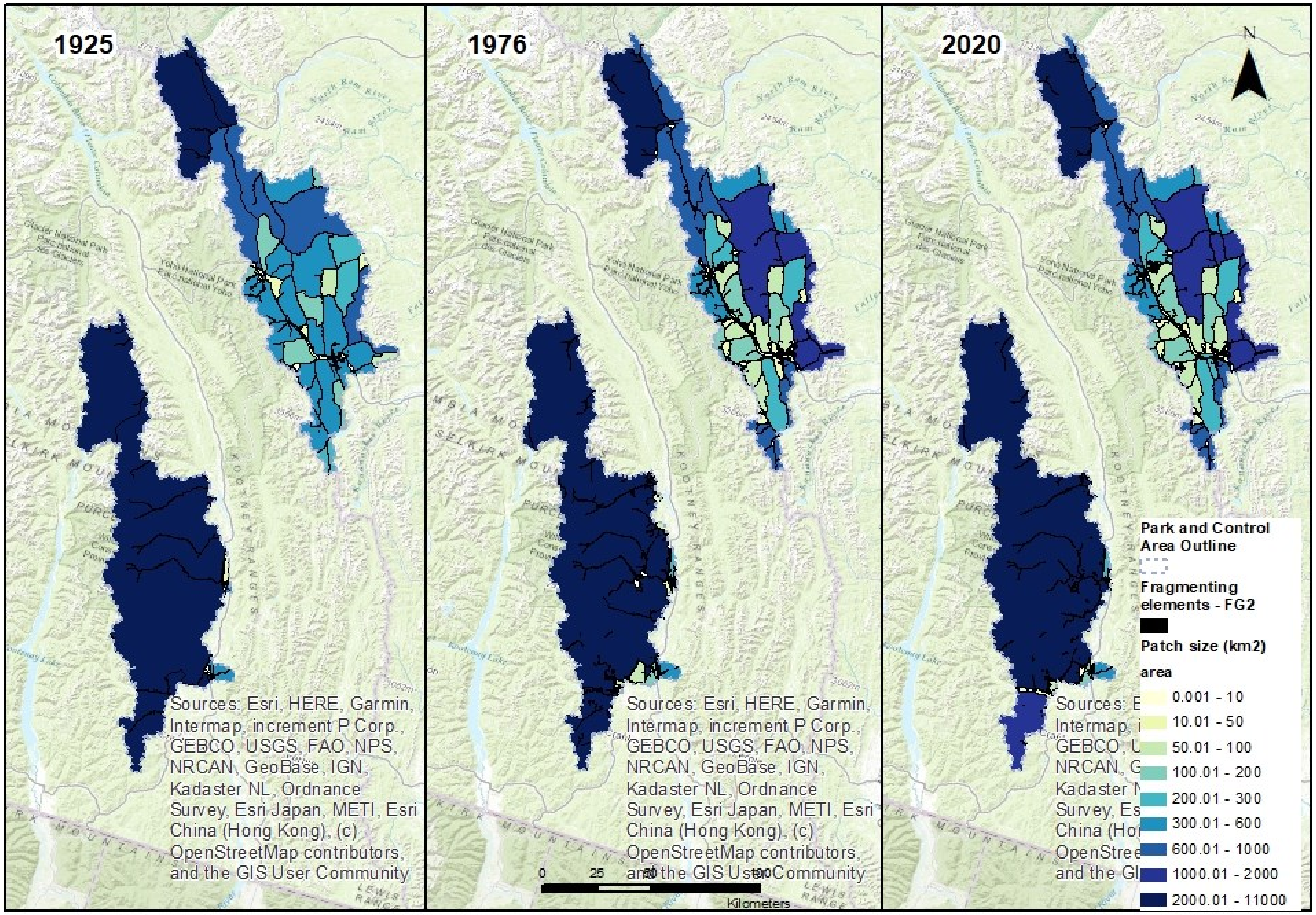


Figure 3.3.1f- *Example: A map showing changes in patch size over time in Banff NP (north-east) and its associated control area. The park has been subject to more fragmentation over time in comparison to its control area.*

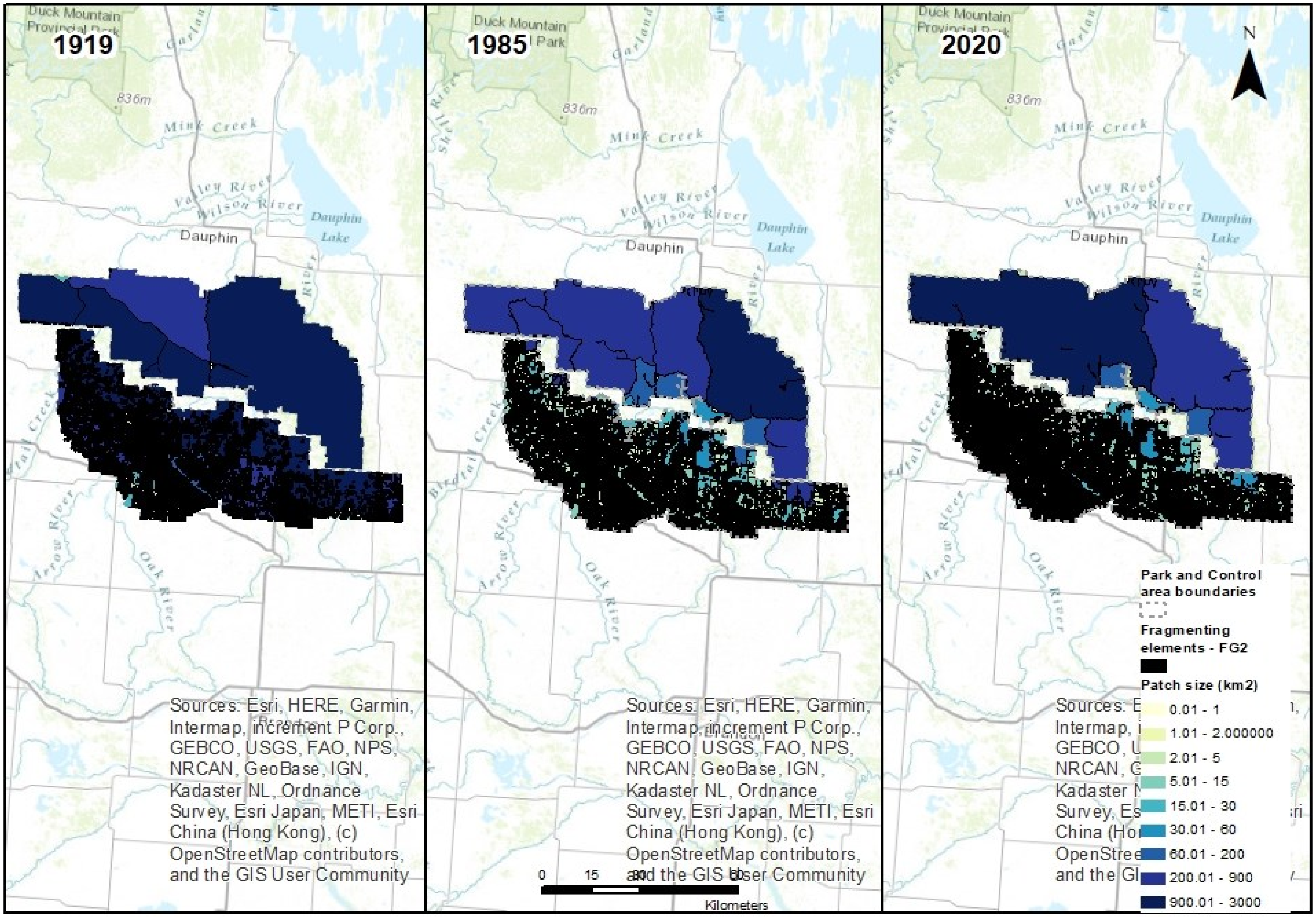


Figure 3.3.1g- *Example: A map showing changes in patch size over time in Riding Mountain NP (north) and its associated control area. The control area has been subject to more fragmentation across all time steps, particularly relating to agricultural land.*

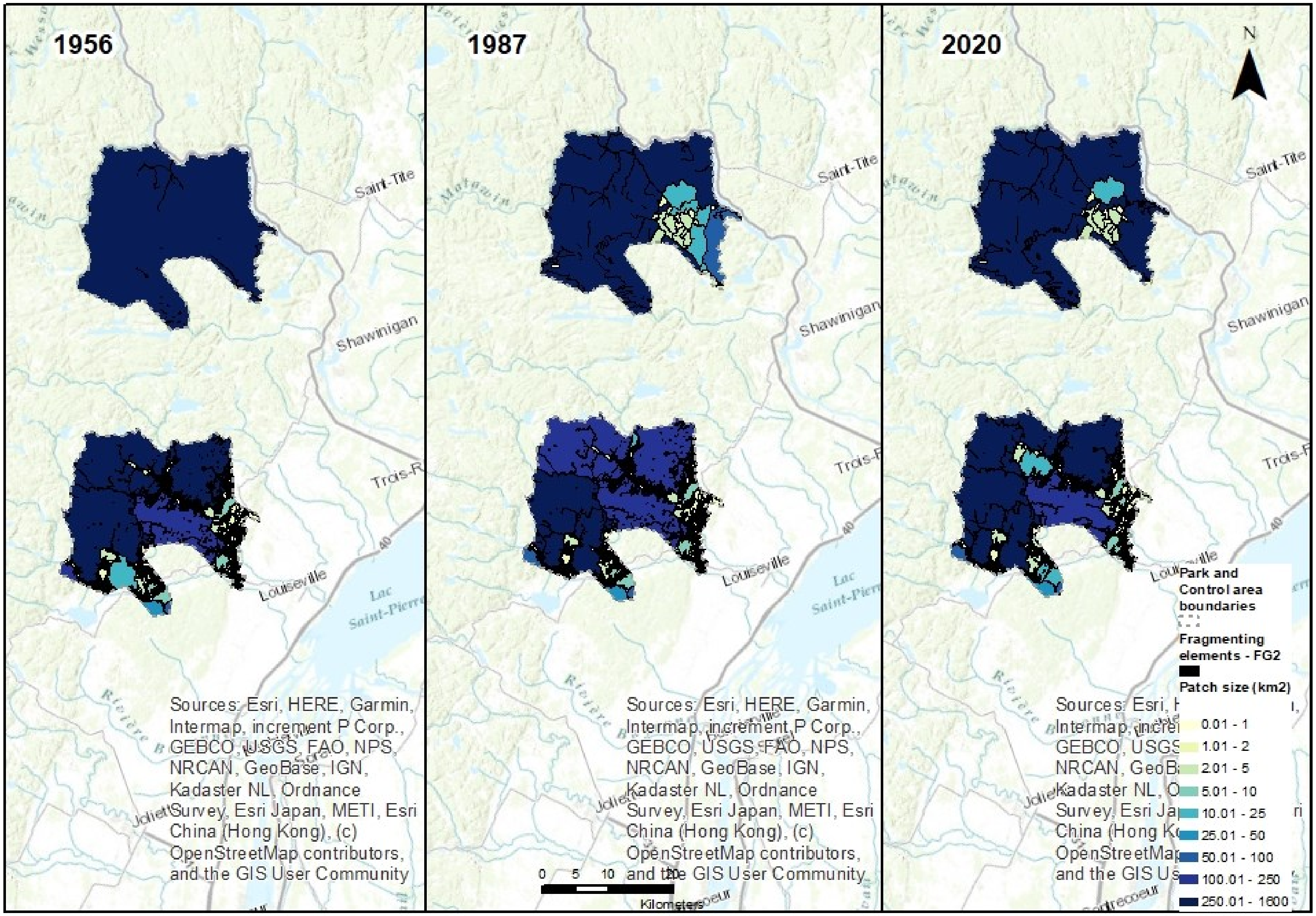


Figure 3.3.1h- *Example: A map showing changes in patch size over time in La Mauricie NP (north) and its associated control area. Although there has been a decrease in effective mesh size over time, the control area has had a greater increase in fragmentation levels.*

3.3.2 National Park System

The results from the PC-BACIPS analysis for the anthropogenic fragmentation geometries (CBC procedure of FG1 and FG2) of the National Park System suggest that overall, park protection has been successful in preventing landscape fragmentation of natural areas, in comparison to similar unprotected control areas. The PC-BACIPS method selected a linear model as the most suitable.

For these control areas, the results in table 3.3.2 for both FG1 and FG2 infer that the National Park System protection has prevented the likely landscape fragmentation related to the anthropogenic features considered in the analysis. The parks and control areas diverged in effective mesh size levels at a rate of 3.47 km^2/yr and 7.18 km^2/yr for FG1 and FG2 respectively, with the control areas decreasing in effective mesh size more than the parks in the same amount of time. However, the insignificance of the results for FG1 suggest that with this sample of control areas, the null hypothesis cannot be rejected for this fragmentation geometry, across the whole park system.

Table 3.3.2*- Results of the PC-BACIPS analysis for FG1 and FG2 (CBC procedure) of the National Park System as a whole, with effect estimate, significance and likelihood of model type selection*

	Effect size (km^2/yr)	*p*-value	Model type
FG1	3.47	0.1	Linear (69.4%)
FG2	7.18	0.02	Linear (91.5%)

Figures 3.3.2a and b show the average simple BACI change (step) from before and after park designation for each park, along with their region. This provides context for the overall effect sizes for FG1 and 2, as the parks in FG2 have a much wider range of BACI change sizes, leading to a stronger linear shape after designation.

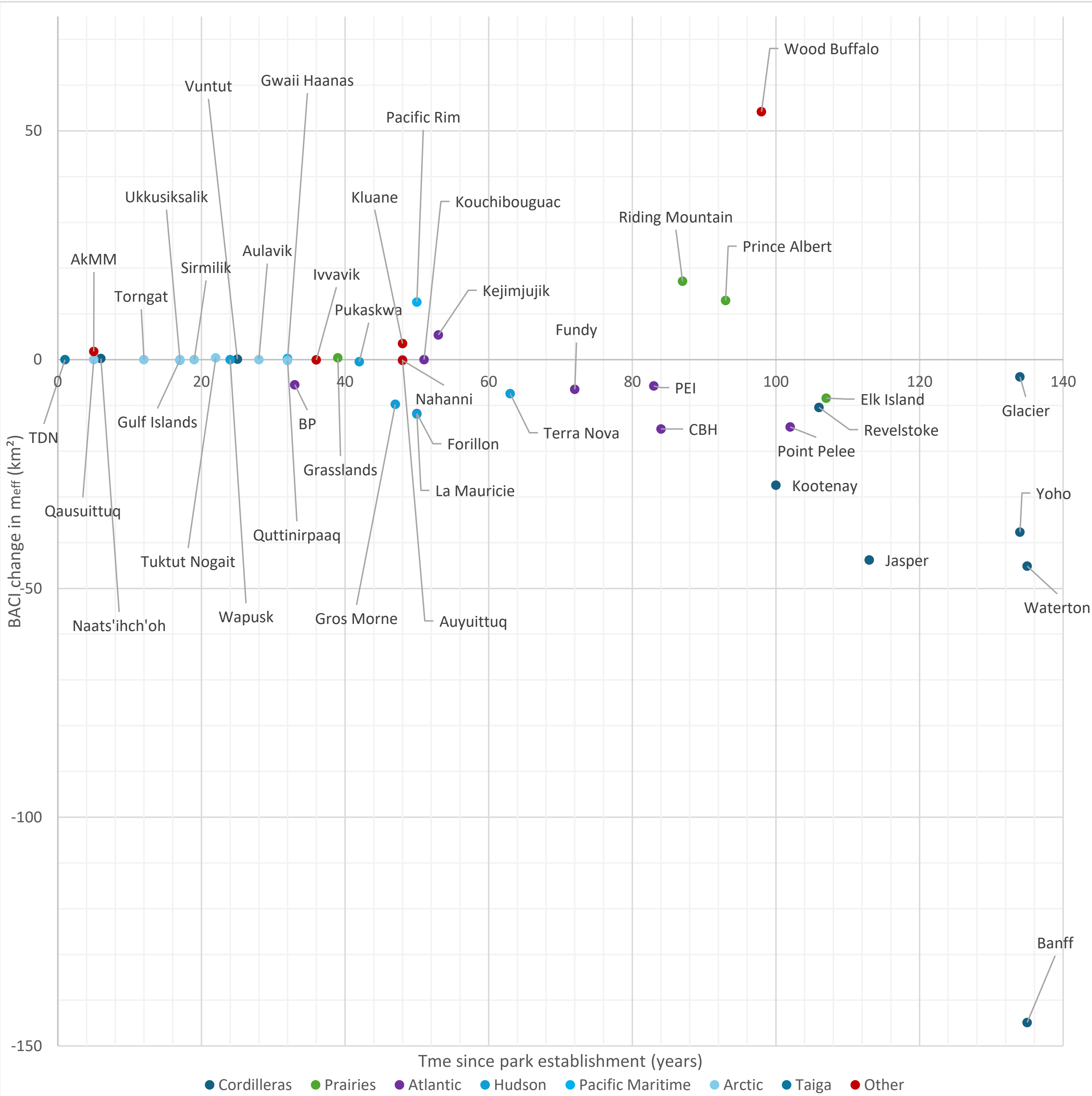


Figure 3.3.2a- *Park effectiveness compared to time since protection, using CBC results for the average BACI change in FG1. A positive BACI change value indicates a stronger increase in fragmentation in the control area over time. A negative BACI change value indicates a stronger increase in fragmentation in the park or park reserve over time.*

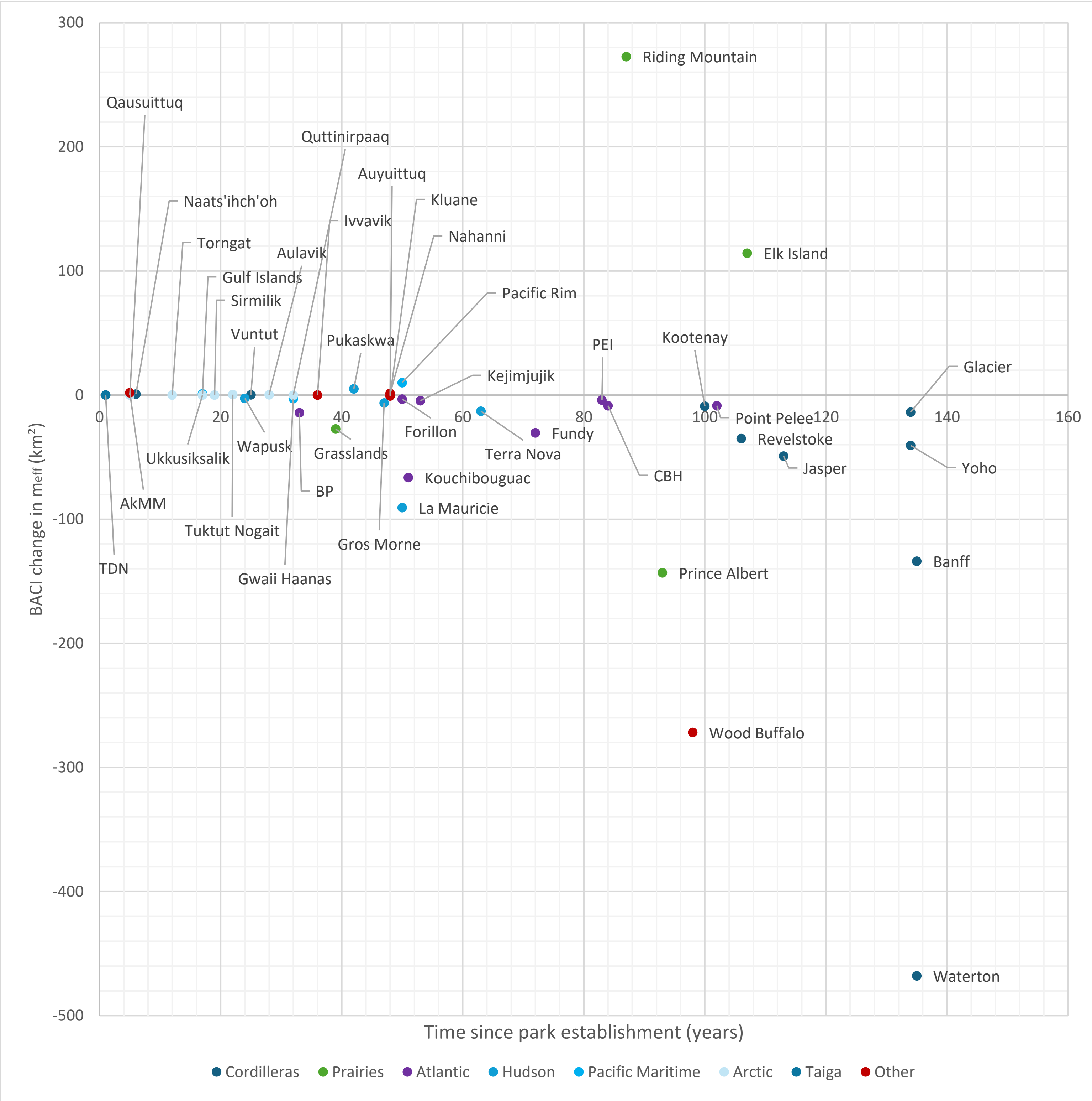


Figure 3.3.2b- *Park effectiveness compared to time since protection, using CBC results for the average BACI change in FG2. A positive BACI change value indicates a stronger increase in fragmentation in the control area over time. A negative BACI change value indicates a stronger increase in fragmentation in the park or park reserve over time.*

3.3.3 Subgroups within the parks system

The PC-BACIPS analysis was run for subgroups of the park-control pairs based on their creation at each management time step, and their regions.

3.3.3.1 Fragmentation levels per management time step

Table 3.3.3.1*- Results of the PC-BACIPS analysis for FG1 and FG2 (CBC procedure) of parks created in each management time step, with effect estimate, significance and likelihood of model type. Dark green shading indicates a significant result.*

Time step		Effect size (km^2/yr)	*p*-value	Model type
1885-1930	**FG1**	-0.266	0.02	Linear (61.8%)
	FG2	-0.661	0.8	Linear (48.1%)
1931-1969	**FG1**	0.206	0.4	Linear (54.6%)
	FG2	11.8	0.3	Linear (41.7%)
1970-1999	**FG1**	1.32	0.8	Linear (50.2%)
	FG2	1.86	0.8	Linear (50.8%)
2000-2018	**FG1**	-6.29	0.9	Linear (47.9%)
	FG2	-6.51	0.9	Linear (48%)
Current	**FG1**	0.0122	2.00E-14	Linear
	FG2	-0.0413	6.00E-15	Linear

As shown in table and figure 3.3.3.1, the largest effect size was found for FG2 in parks designated between 1931 and 1969: a divergence in effective mesh size of 11.8 km^2/yr. This indicates that when considering all anthropogenic fragmenting elements, these parks have been successful in preventing landscape fragmentation in comparison to their control areas. The time-step for parks designated between 1970 and 1999 also has a positive effect size.

For the first parks designated before 1930, the park "protection" has not been successful in preventing landscape fragmentation from the elements considered in FG1 and FG2, albeit to a small degree. For these parks, there has been a minimal divergence in fragmentation levels of -0.27 km^2/yr and –0.66 km^2/yr between the parks and control areas, with parks decreasing in effective mesh size more than in the control areas. The most recent time-step for park designation suggests that with the control areas considered in this analysis, park designation has also not been successful in preventing anthropogenic landscape fragmentation, with a divergence in effective mesh size of over -6 km^2/yr.

The results for FG2 in parks designated before 1930, and for FG1 and FG2 up to 2018, are insignificant. It cannot be determined whether park protection elsewhere at these time-steps would have generally prevented increasing fragmentation over time for anthropogenic fragmenting elements. This could be due to a large variation within these time-step sub-groups, or a too small sample size - particularly for the 1931-1969 and 2000-2018 sub-groups.

The results for the current time-step are significant due to having only one park in this time-step – Thaidene Nëné.

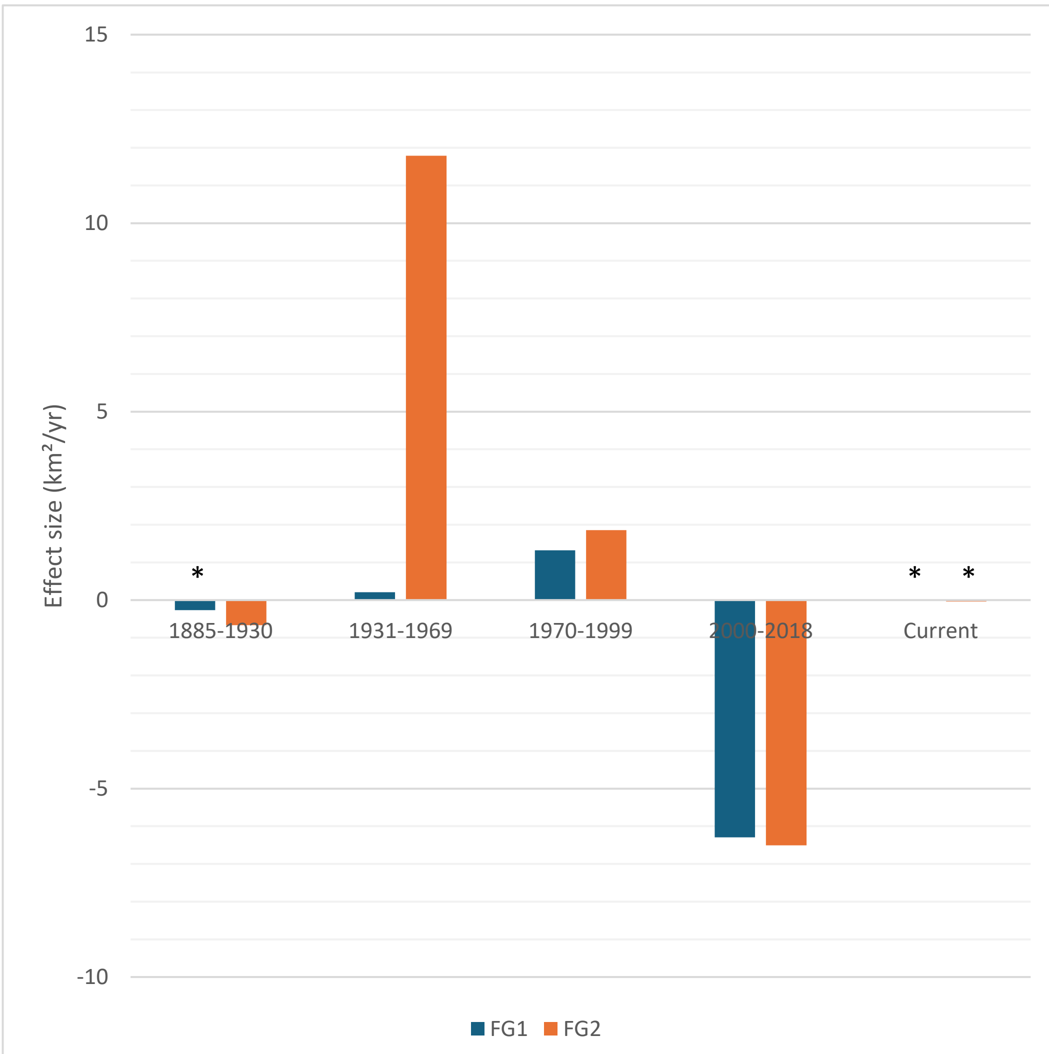


Figure 3.3.3.1- *A bar chart demonstrating the effectiveness of parks created within each management time-step (* = $p < 0.05$).*

3.3.3.2 Fragmentation levels per region

Table 3.3.3.2- *Results of the PC-BACIPS analysis for FG1 and FG2 (CBC procedure) of parks in each region, with effect estimate, significance and likelihood of model type selection*

Time-step		Effect size (km^2/yr)	*p*-value	Model type
Cordilleras	**FG1**	1.95	0.5	Linear (51.8%)
	FG2	2.62	0.4	Linear (55.9%)
Prairies	**FG1**	1.1	0.1	Linear (63%)
	FG2	1.69	0.8	Linear (48.6%)
Atlantic Maritime & Great	**FG1**	-0.144	0.2	Linear (61.3%)
Lakes	**FG2**	-0.909	0.1	Linear (73.1%)
Hudson	**FG1**	8.59	0.06	Linear (79.6%)
	FG2	9.23	0.1	Linear (73.1%)
Pacific Maritime	**FG1**	-12.4	0.4	Linear (57.1%)
	FG2	-12.4	0.4	Linear (56%)
Arctic	**FG1**	-4.3	0.9	Linear (48.6%)
	FG2	-4.3	0.9	Linear (48.6%)
Taiga	**FG1**	5.5	0.2	Linear (66.4%)
	FG2	25.5	0.04	Linear (81%)

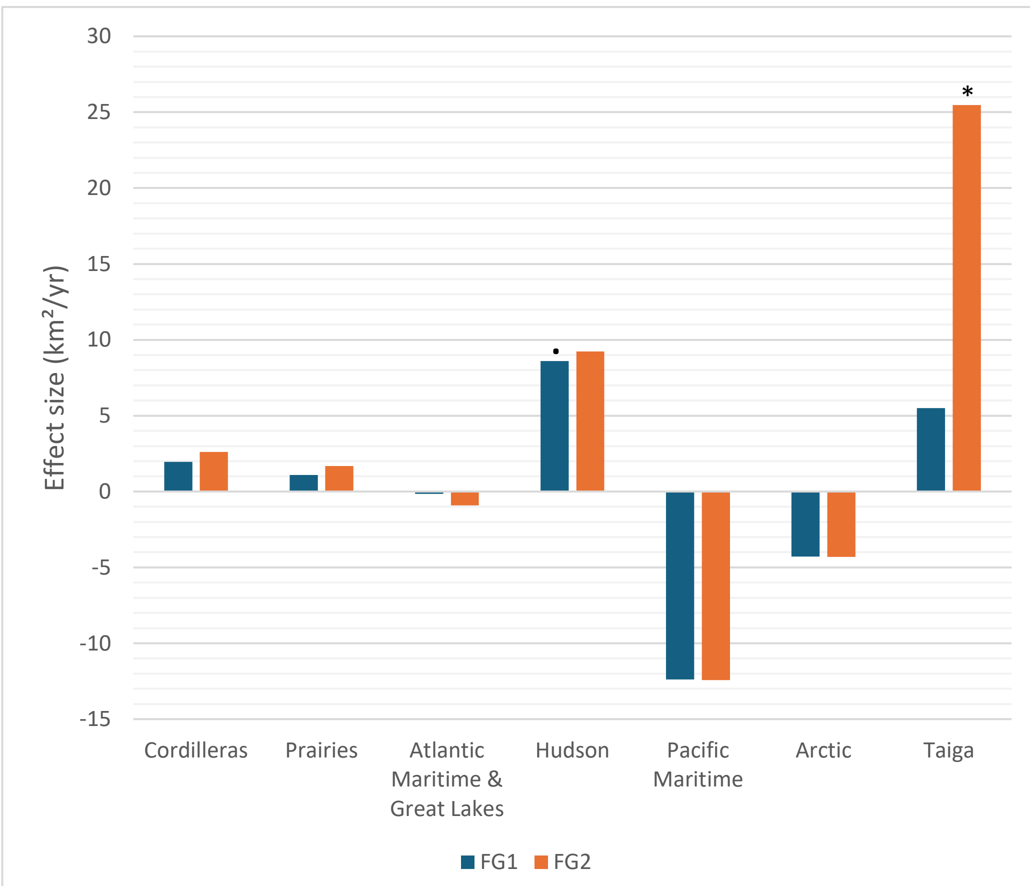


Figure 3.3.3.2- *A bar chart demonstrating the effect size of parks per region for each fragmentation geometry (* = p < 0.05; • = 0.1 > p > 0.05).*

Table and figure 3.3.3.2 demonstrate that most significantly, there was a divergence of 25.5 km^2/yr between the parks and control areas of the Taiga region for FG2, meaning that the parks in this area were successful in preventing the likely fragmentation from all anthropogenic fragmenting elements considered in this study. Similarly, the parks in the Hudson region were successful in preventing fragmentation associated with FG1 and FG2, followed by parks in the Cordilleras and Prairies regions.

The least successful region for parks preventing the expected fragmentation levels of the control areas was the Pacific Maritime, with effect sizes of over -12 km^2/yr. The Arctic parks were also unsuccessful in preventing landscape fragmentation in comparison to their control areas.

For the Atlantic Maritime and Great Lakes, it cannot be inferred that the parks and control areas differ in effective mesh size over time based on their geographic location, owing to the small effect size.

3.4 Discussion

3.4.1 Standalone park-control pairs

The PC-BACIPS analysis revealed that there is a divergence of effective mesh size between the park and control area in a negative or neutral direction for a majority of the park-control pairs, indicating that park protection has only been somewhat successful in preventing the likely change in effective mesh size that has occurred in the unprotected control area.

3.4.1.1 Park-control pairs where park designation has been unsuccessful in preventing landscape fragmentation

Of note, compared to its control area, Banff NP has been subject to a much higher level of fragmentation overall since 1925 (before its conservation through the National Parks Act of 1930). The effective mesh size has steadily decreased over the last century in the park across all fragmentation geometries, but the greatest effect size (and significance) was seen in FGs 1, 2 and 3. The direction of the effect is negative, demonstrating that park "protection" has not prevented fragmentation. This may be due to Banff's status as Canada's first national park, where it was originally developed with tourism in mind, rather than ecological integrity. Between 1950 and 1974, the significant decrease in effective mesh size is likely due to the construction of highway 1 through the park. In addition, there has been expansion in campgrounds, the trail system, and ski areas. However, in recent years there has been a focus on reducing the footprint of heavily trafficked areas, including decommissioning trails and facilities in high-quality habitat (Parks Canada, 2022c) and lease reductions of the ski areas (Parks Canada, 2017c). In contrast to Banff NP, the control area has decreased its fragmentation slightly since 1974. Based on visual inspection of the input data, this appears to be due to a decrease in productive agricultural area since this time, and a decrease in maintained trails and unpaved roads likely related to forestry or coal mining (Ministry of Forests, Lands, Natural Resource Operations and Rural Development, 2020), thus theoretically removing barriers to animal movement.

The CUT and CBC procedures demonstrate similar patterns for La Mauricie as for Banff, indicating that there are similar changes in fragmentation both within the park and control area boundaries over time, but for different reasons. Effective mesh size remained stable for FG1 in the control area. However, effective mesh size for FG2 increased by 2012 and 2020, due to a decrease in productive agricultural area to the southwest of the control area. In comparison, the effective mesh size of FG1 and FG2 decreased in the park in 2020 due to an increase in tourist facilities such as oTENTik camping infrastructure and picnic sites (Parks Canada, 2023). Similar patterns can be seen in the Kouchibouguac and Terra Nova park-control pairs, with smaller effect sizes.

Overall, there is little change in fragmentation levels over time for either Yoho NP or its related control area, but across most FGs the park was more fragmented than the control area. This was due to the original development related to the railway line and subsequent development outwards, in comparison to the undeveloped backcountry of the control area. A deviation from this trend is seen in the FG3 CBC results, where the park has a larger effective mesh size than the control area over time. This results from there being fewer natural fragmenting elements (such as water or ice) in the immediate buffer zone of the park than that of the control area. The effect size is still negative, due to a decrease in effective mesh size over time in the park but not so much in the control area. Similar trends can be seen for other Rocky Mountain park-control pairs

– Glacier, Jasper, and to an extent, Mount Revelstoke. This is due to their shared history with the Canadian Pacific Railway, subsequent tourism development, and early lumbering and mining within park boundaries (Dearden & Berg, 1993; Lothian, 1987; McNamee, 2010).

Another park-control pair where park designation has been unsuccessful in preventing landscape fragmentation is Wood Buffalo. While there was a small positive effect size for FG1 in the CUT and CBC procedure due to a greater decrease in effective mesh size over time in the control area, FG2-FG5 a demonstrate a divergence of CBC effect sizes of over $-3km^2/yr$. The anthropogenic fragmentation level in the park remained relatively stable, in comparison to the control area where there was an increase in effective mesh size in 2011 and 2020 after a slight decrease from 1919-1986. This was due to a decrease in recorded productive agricultural land around the town of High Level.

3.4.1.2 Park-control pairs where park designation has been successful in preventing landscape fragmentation

The size for the Point Pelee park-control area was small, which does not demonstrate the stark difference in effective mesh size between the park and control area in all FGs. The national park was the first park to be established with conservation as the foremost reason, in 1918 (Parks Canada, 2018b). Overall, the park has prevented the encroaching urbanisation of the Greater Toronto Area. However different the park and control area are, the overall trends in fragmentation levels looks very similar between the two in that there was a decrease of effective mesh size until 2010, when there was an increase for FG2-FG5. This was mostly due to the removal of some productive agricultural land.

A park-control pair that demonstrated a significant positive effect from before designation to after is Riding Mountain. Here, the effective mesh size in the park stayed stable for FG1 of the CUT and CBC procedure. For FG2 of CBC, there was a slight decrease in effective mesh size in 2012 and 2020, due to an increase in unpaved roads in the 10 km buffer around the park. In comparison, FG2 in the control areas demonstrated a larger decrease in effective mesh size in 2012 and 2020 due to an increase in unpaved roads and agriculture in the area. Across all time-steps, the control area for FG2-FG5 was far more fragmented than the park.

This agrees with the conclusion that Canadian prairie landscapes have been heavily fragmented over time (Roch & Jaeger, 2014), and demonstrates that protection of land like in Riding Mountain NP prevents landscape fragmentation from human influence such as roads in an increasingly suburbanized landscape. The increase in fragmentation in more recent time steps also suggests that in this part of the Canadian prairies, fragmentation in unprotected areas has further worsened since the recommendations made by Roch & Jaeger in 2014.

The trends in effective mesh size over time demonstrate that the Grasslands park-control pair has similar patterns as Riding Mountain in that the control area is much more fragmented than the park. However, both the park and control area at Grasslands had increasing effective mesh size for almost all fragmentation geometries, and for both the CUT and CBC procedure. This indicates a decrease in fragmentation over time, a surprising result for a prairie ecosystem. This due to a decrease in unpaved roads and cart tracks over time, giving way to more cleanly-parcelled land for agriculture. Overall, it appears that in terms of agriculture and its related unpaved tracks/roads, the park designation has been successful in reducing landscape fragmentation and increasing effective mesh size, and landscape fragmentation has decreased over time in the associated control area too.

3.4.1.3 Park-control pairs where park designation has been neutral in preventing landscape fragmentation

In Pukaskwa park-control pair, there was little change in fragmentation over time, and minimal difference in effective mesh size between the park and control area. There was a small increase in trails and unpaved roads in both the park and control area. Its location in Northern Ontario means that the low-population density and highly forested land cover (Ontario Biodiversity Council, 2021) in the area already seems to prevent a significant amount of anthropogenic fragmentation, demonstrating how remote parks often have lower rates of fragmentation. The main difference between the park and control area was seen in FG3 and FG5 due to higher natural fragmentation in the control area from watercourses and waterbodies.

For both the CUT and CBC methods in Wapusk, there was little difference between the park and control area for FGs 1 and 2. This is likely due to the lack of human infrastructure in

Wapusk and surrounding areas, with the exception of research camps (built in the late 2000s), and derelict mining and fur trading facilities, such as York Factory and Port Nelson (Parks Canada, 2024). There was little change over time, due to a lack of human development in Wapusk, the neighbouring Churchill Wildlife Managment Area, or in the tundra to the south-east of the park. In fact, the control area saw a very small decrease in fragmentation around the ruins of York Factory and in the north-east corner of the Wildlife Management Area, due to old rail-lines and ATV tracks becoming overgrown. Tundra buggies are now the main form of transport in the park and control area, which are not overly-damaging to the tundra environment as they travel on already-established trails (HydraForce.com, 2022). FG3 showed that there was more fragmentation from natural barriers in the control area than in the park.

In the northern parks of Auyuittuq, Nahanni, Ivvavik, Quttinirpaaq, Aulavik, Vuntut, Sirmilik, Ukkusiksalik, Torngat Mountains, Nááts'įhch'oh, Qausuittuq and Thaidene Nëné, the CUT and CBC results showed minimal divergence in fragmentation levels since before park designation to after. Most of the human-induced fragmentation was done in the form of research stations, airstrips and remote campsites. The current lack of human development in Canada's northern regions and the more recent park designation dates for the majority of these pairs can explain this lack of change in effective mesh size. Their remoteness also explains why their effective mesh size is much lower when considering natural barriers since their landscapes are inherently fragmented by water and ice (Wulder et al., 2011). However, it is expected that industries relating to natural resource extraction – oil & gas, mining, forestry, fisheries – and potentially the emerging tourism industry, will grow in the North in coming years due to a push for economic development and climate change affecting operations further south (Southcott, 2009; Conference Board of Canada, 2010). This region has large areas of unprotected natural landscape that are under threat from industrial disturbances that could affect local wildlife such as caribou (UNESCO World Heritage, 2006; Stewart et al., 2020). Overall, a lack of anthropogenic fragmentation in northern control areas and in park buffer zones at present day does not suggest that there will not be any future industrial development in these areas later on. For example, the Prairie Creek zinc mine site is in the buffer zone of Nahanni NPR. It has not yet operated, but the mining company has recently built an all-season road through the park (Mining.com, 2023) and the site has been described as having "the potential to impact the ecological integrity and cultural resources” of the local watershed and the park reserve (Parks Canada, 2021c:8).

3.4.1.4 CUT and CBC procedure: fragmentation near the boundaries of national parks

Overall, the CBC procedure is more appropriate for this study because it does not consider the boundaries of the parks and control areas as physical barriers. This is particularly useful for inland parks that have multiple sections to them, such as Grasslands and Nááts’įhch’oh. However, the CUT and CBC results can be compared to gauge the effect of fragmentation adjacent to the parks and park reserves. Studying the land adjacent to protected areas gives an idea of how a park can be affected by processes that occur outside its boundaries, and how isolated the wildlife populations might be from any surrounding natural area (Soverel et al., 2010; Parks Canada 2023). Pairs that have a greater negative effect size for the CBC procedure than the CUT indicate that there is more fragmentation in the 10 km buffer zone of the park in comparison to within the park boundary, and vice versa, since the divergence of effective mesh size can be compared between the CUT and CBC. This is the case for almost all park-control pairs with the exception of Prince Albert (Figure 3.4.1.4a). The fragmentation level change is greater within the boundaries of Prince Albert NP than in the buffer zone. The parks with the strongest indication of increased fragmentation just outside their boundaries in the greater park ecosystem are primarily in south-eastern Canada: Kouchibouguac, La Mauricie, Gros Morne, Pukaskwa, and Bruce Peninsula. This generally follows the findings of Leroux & Kerr (2013), that small park sizes and proximity to urban areas in the south of Canada encourage development close to and into the boundaries of protected areas.

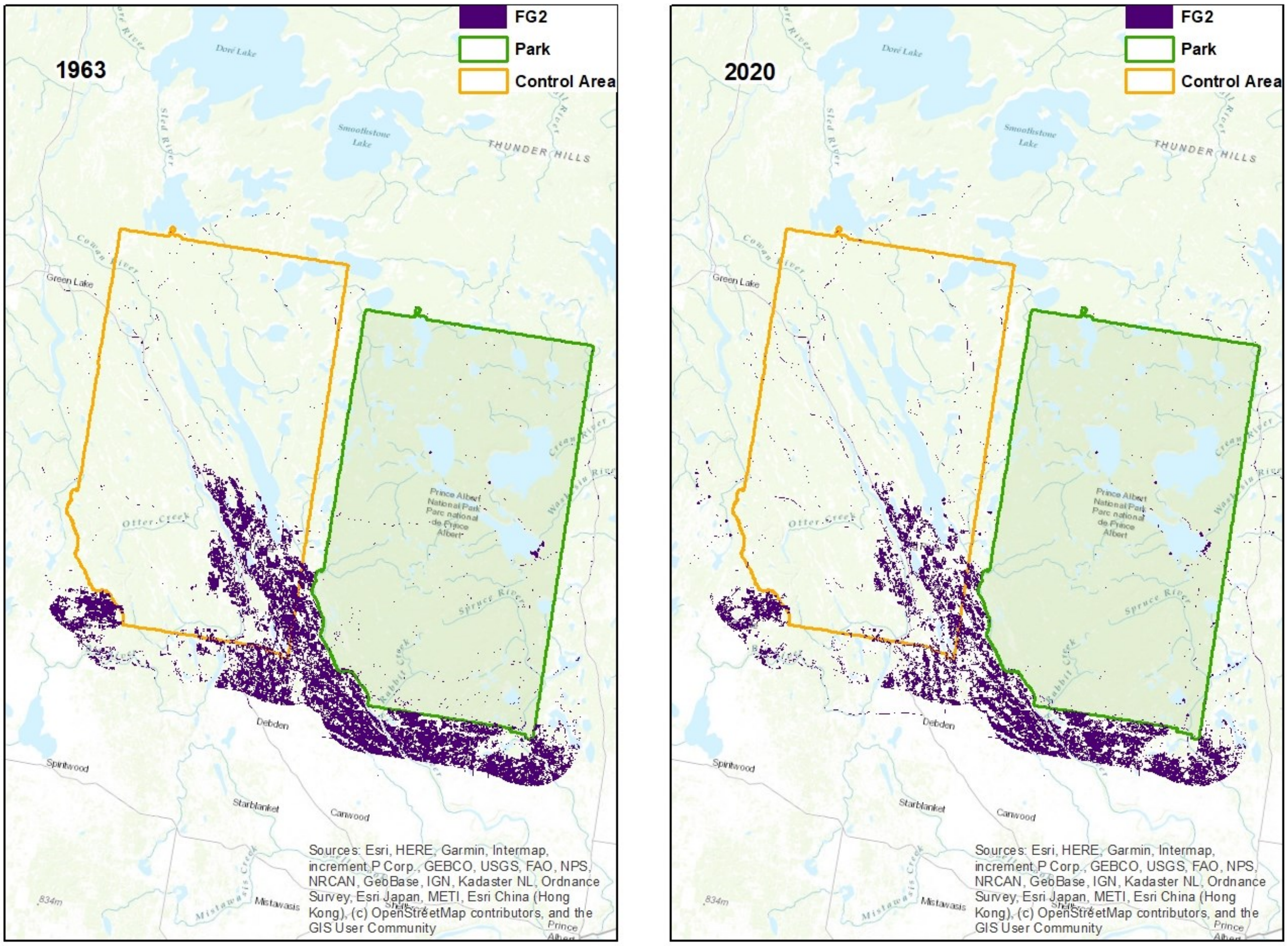


Figure 3.4.1.4a- *FG2 for Prince Albert NP and its associated control area, in 1963 and 2020. There is a greater reduction of fragmenting barriers in the park's buffer zone than within its boundaries, along with a large reduction of fragmenting barriers in the control area and its buffer zone. Therefore, the effect size for the CUT procedure (-0.874 km²/yr) is larger than for the CBC (-0.816 km²/yr).*

For the majority of park-control pairs, trends in effective mesh size change for both the CUT and CBC procedures were similar, indicating that fragmentation around the parks may increase pressure for fragmentation and development within the park boundaries (Leroux & Kerr, 2013). However, the Pacific Rim park-control pair (Figure 3.4.1.4b) has a negative direction of effect size for the CUT procedure, and positive for the CBC procedure. The CUT results show that Pacific Rim NPR has had more fragmentation over time than its control area, and vice versa with the CBC results. This is due to an increase in tourist facilities within Pacific Rim NPR but not adjacent to its boundaries, and a significant increase in the suburban built-up areas of Victoria in the buffer zone of the control area in comparison to the smaller effect of the expansion of Tofino and Ucluelet near the park.

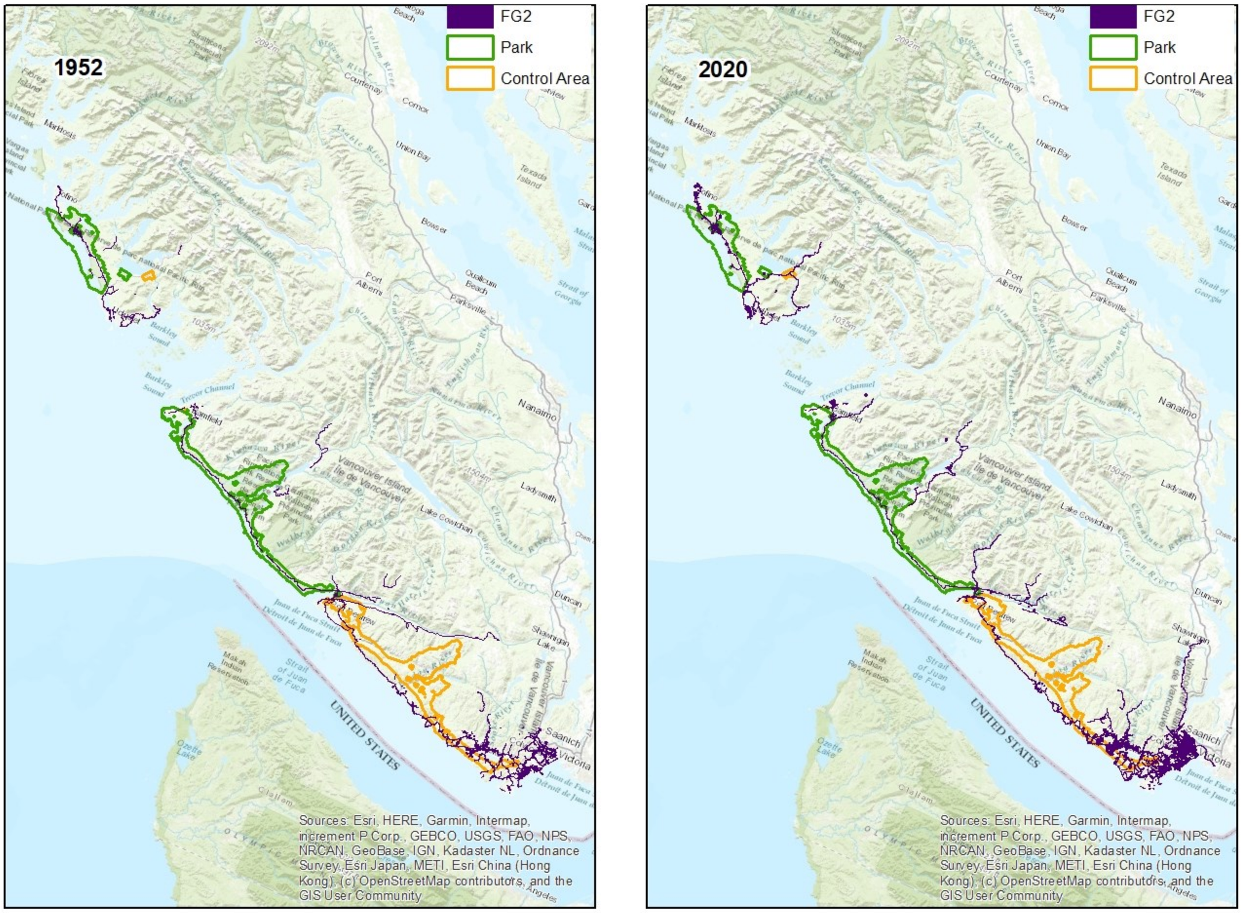


Figure 3.4.1.4b- *FG2 for Pacific Rim NPR and part of its associated control area, in 1952 and 2020. There is a greater increase of fragmenting barriers in the NPR than in its buffer zone, and a greater increase of fragmenting barriers in the buffer zone of the control area in comparison to within its boundaries. Therefore, there is a negative effect size for the CUT procedure (-0.0437 km ^{2}/yr) and a positive effect size for the CBC (0.168 km ^{2}/yr).*

3.4.1.5 Examples: Keystone species and species at particular risk from fragmentation, and their history in the park-control pairs

Many parks have keystone species significantly affected by landscape fragmentation in the protected areas and greater park ecosystems of the park buffer zones and control areas. For example, Banff has undertaken a Plains Bison Reintroduction Pilot, to support Parks Canada's goals in maintaining and restoring ecological integrity. Free roaming plains bison (*Bison bison*

bison) became locally extinct in Banff by the 1870s due to overhunting, but are important to the native ecosystem of the park as their presence affects the landscape in ways that benefit other species (Parks Canada, 2018a; Parks Canada, n.d.). The new bison herd movement has been influenced by interventions such as drift fences in order to keep the herd within their reintroduction zone, however their movement in the first five years of reintroduction shows their attraction to areas outside of Banff NP at Ya Ha Tinda and the Panther River valley (Parks Canada, 2023c). Keeping landscape fragmentation low in and around the eastern slopes of Banff NP will be crucial in future range and population management for the herd. A similar reintroduction program for plains bison has occurred at Grasslands NP, with the herd originating from Elk Island NP. Elk Island is the only entirely fenced national park, in order to maintain the disease-free status of the plains bison and wood bison (*Bison bison athabascae*) population. This fencing, and the traffic on Highway 16 split the park into two blocks that are ecologically independent of one another and independent of the surrounding buffer zone, and it contributes to the park's challenges of overabundant bison, elk (*Cervus canadensis manitobensis*) and moose (*Alces alces andersoni*) (Parks Canada, 2023e).

Another keystone species that is native to a wide range of park-control pairs is the caribou. The caribou's keystone status and its sensitivity to landscape fragmentation and human disturbance led to the creation of the 10km buffer zone for the parks and control areas in the present study. Logging, road development and associated industrial and recreational access bring increased hunter access, and altered predator-prey balances have displaced Mountain, Boreal, Atlantic and Newfoundland caribou (*Rangifer tarandus caribou*), affected their habitats and retracted their ranges (Adamczewski et al., 2003; Apps & McLellan, 2006; Arlt & Manseau, 2011). The study shows the increasing landscape fragmentation and human pressures on the Rocky Mountain national parks and eastern national parks, which affect caribou access to undisturbed ranges. Actions taken by Parks Canada to reduce harm to caribou include mitigation measures and closures of roads, recreation areas, and trails in Gros Morne, Jasper, Glacier and Mount Revelstoke (Parks Canada, 2023f). The Peary Caribou (*Rangifer tarandus arcticus*) historical habitat that includes Quttinirpaaq, Qausuittuq and Aulavik is essentially all available and has not been lost or fragmented by anthropogenic developments (COSEWIC, 2015). This can be seen by the small effect sizes in both the CUT and CBC procedures for these parks. This is also mostly the case for other Barren-ground caribou, with their range covering Ivvavik, Vuntut, Tuktut Nogait, Thaidene Nënë, Ukkusiksalik, Sirmilik and Auyuittuq.

Mammals are not the only animals affected by landscape fragmentation in the park-control pairs. The wood turtles (*Glyptemys insculpta*) native in and around La Mauricie NP are particularly sensitive to the human-induced landscape fragmentation that has increased over time, and its status in the park is considered poor (Parks Canada, 2024a). The southern part of the greater park ecosystem contains increasing urbanization concentrated around Trois-Rivières (Habitat, 2022). Due to these fragmenting barriers encroaching into the wood turtle's habitat in the control area and the park buffer zone, actions taken by Parks Canada in the park include installing structures to guide turtles off the road and plant rehabilitation to make medians less attractive to turtles (Parks Canada, 2024a).

3.4.2 National Park System as a whole

The PC-BACIPS analysis of FG2 for the entire NPS gives an overall average divergence in effective mesh size of 7.18 km^2/yr since park protection dates. The control areas have increased in fragmentation levels in comparison to the parks, which indicates that the NPS has been successful in curbing the fragmentation that is expected in unprotected areas of a similar landscape and in this way has been improving the ecological integrity in national parks and park reserves, or has helped slow down deterioration.

3.4.2.1 Landscape fragmentation as part of ecological integrity monitoring in the NPS

Not all parks monitor landscape fragmentation and/or connectivity, directly or indirectly through other ecological indicators, and those that do use a variety of methods. As the NPS encompasses such a wide range of environments, there is no one-size-fits-all method for conservation. Parks that explicitly include landscape fragmentation, or indicators related to it, in their ecological integrity monitoring include long-established Rocky Mountain parks and Atlantic coastal parks (table 3.4.2.1). It is unknown whether any other parks are using landscape fragmentation as part of their monitoring to understand the rate of landscape change (OAGC, 2013). Some parks monitor some form of connectivity- or fragmentation-related indicators through projects that are part of the Conservation and Restoration Programs, such as the wildlife crossing projects in Kootenay and Bruce Peninsula national parks (Parks Canada, 2022). However, this monitoring does not always appear to be part of the parks' official EIM, leading to their lack of inclusion in State of Park reports and other management documents.

Table 3.4.2.1 *- Parks that consider landscape fragmentation or related indicators in their ecological integrity monitoring, as found in the most recent State of Park reports or Park Management Plans.*

Park	Indicator
Banff	Winter wildlife corridors
Waterton Lakes	Sensitive species-secure habitat
Jasper	Motorized access density
Mount Revelstoke	Wildlife cameras & wildlife tracks
Point Pelee	Forest ecosystem monitoring
Kootenay	Wildlife cameras & wildlife tracks
Fundy	Carnivore habitat connectivity
Terra Nova	Wildlife cameras
La Mauricie	Forest landscape monitoring, black bear movement corridors
Pacific Rim	Wildlife habitat fragmentation & anthropogenic development
Bruce Peninsula	Forest & wetland connectivity & abundance
Wapusk	Maintaining natural physical processes

Despite the high levels of intactness in the Arctic and sub-Arctic regions, ecological integrity cannot be adequately considered from just a landscape fragmentation metric. The dominant drivers of change are indirect stressors such as climate change and ice pack melt, and measuring these as ecological indicators are better suited for more northern environments (COSEWIC, 2015; Trammel et al., 2022). Therefore, it is logical that anthropogenic-related landscape fragmentation is not currently considered in ecological integrity monitoring in many northern parks.

3.4.2.2 Landscape fragmentation monitoring within the greater park ecosystems

For the greater park ecosystems (including unprotected land adjacent to the protected areas and control areas), landscape fragmentation monitoring is important but missing in many regions. Remaining natural land cover between protected areas are vulnerable to ongoing landscape fragmentation, as seen in many of the control areas in this study, leaving the parks and park reserves as increasingly isolated stepping-stones of natural areas within a wider human-influenced landscape (MacArthur & Wilson, 1967; Roch & Jaeger, 204; Cole et al., 2023). To combat this process, in some parks, Parks Canada is working with other organizations to connect the surrounding landscape. One of the most significant of these is the Landscape Resiliency Program, a collaboration with the Nature Conservancy of Canada, where buffer zones and wildlife corridors adjacent to national parks will be protected by working with local communities. Parks included in this program are Gulf Islands, Waterton Lakes, Grasslands, Bruce Peninsula, Point Pelee, La Mauricie, Kouchibouguac, and Kejimkujik (Nature Conservancy of Canada, 2023).

Other NGOs such as the Y2Y and Staying Connected Initiatives look to protect large swathes of intact habitat, with the NPS acting as cornerstones of the connected landscapes (Staying Connected Initiative, 2023; Yellowstone to Yukon Conservation Initiative, 2024). By focusing on conservation with the wide variety of stakeholders and levels of governance present in a multi-functional landscape outside of a park, these initiatives find success in preventing landscape fragmentation both within and between protected areas (Parrott et al., 2019). Additionally, some provinces are working on monitoring fragmentation and/or connectivity. For example, the government of Alberta and the Alberta Biodiversity Monitoring Institute developed an indicator based on Equivalent Connected Area (Alberta Government, 2024). In Quebec, the Quebec Ecological Corridors Initiative champions ecological corridors in land use planning and advises governments, farmers, and other stakeholders (Nature Conservancy of Canada, 2017).

3.4.2.3 Canada's targets for increasing coverage of protected areas

The long-term goal of the NPS of Canada is to establish "at least one national park in each of Canada's terrestrial regions" (McNamee, 2010; Parks Canada, 2021d). This goal is 79% complete and 31 of the country's 39 terrestrial regions are represented. As shown in figures 3.3.2a and b, individual park effectiveness in preventing landscape fragmentation in the well-represented cordilleras region is low – fragmentation is higher in the park than in the control area since park designation. In comparison, park effectiveness for the prairies and pacific maritime regions has been high despite less representation in the NPS. Despite certain regions being well-represented by national parks, they are not necessarily well-connected or have a history of being well-managed. Of note, Parks Canada states that they are working to maintain or restore the ecological integrity of national parks, emphasizing that some older parks have had their ecological integrity degraded (Parks Canada, 2022c).

As well as completing the NPS so that 100% of Canada's regions are represented, the federal government has set the 30-by-30 target – conserving 30% of Canada's land and water by 2030, in response to the Global Biodiversity Framework's same goal (ECCC, 2022). The predecessor of 30x30 (Target 3) was Aichi Target 11 – the protection of at least 17% of terrestrial and inland water areas and 10% of coastal and marine areas globally – which was successful in quantity but not in quality as many of the areas protected were not adequately connected nor were the most crucial areas for biodiversity conservation (Convention on Biological Diversity, n.d. & 2020; Lo & Jang, 2022). The GBF 30x30 target explicitly includes goals for new protected areas to be effectively managed and well connected (WWF & IUCN, 2023). For improved connectivity and lower levels of fragmentation, some regions in the south of Canada where human impacts are higher will require both protection and ecological restoration of degraded habitat, which has been acknowledged by Parks Canada (Currie et al., 2022; Parks Canada, 2022c). In addition, it is unlikely that a "representative and well-connected network" of conserved land can be achieved with only protected areas, managed by Parks Canada or other agencies (Nature Conservancy of Canada, 2023). Other Effective Area-based Conservation Measures (OECMs) could be considered as part of Canada's path to 30x30 as by definition they are already conserving biodiversity. However, OECMs tend to have a focus on maintaining the integrity of an ecosystem with their current use, which makes restoring degraded ecosystems more difficult (Dudley & Stolten, 2023). Overall, despite the federal government's goals to represent all regions in the NPS and protect 30% of Canada's land and water by 2030, it is not guaranteed that this target will be met if a "business-as-usual" approach is taken, especially with respect to preventing further increases in landscape fragmentation.

3.4.2.4 Indigenous management and co-management of protected areas

Concerns have also been raised regarding Canada's focus on partnerships with Indigenous peoples through Indigenous Protected and Conserved Areas (IPCAs) and its efficacy in meeting the 30x30 target. The stewardship of protected areas disproportionately falls on Indigenous communities, particularly in the north, often with poor plans for long-term management (Currie et al., 2022). Canada's funding model has been deemed insufficient by many communities and a lack of provincial support has caused tension over land titles (Cruickshank, 2022; Kwetásel'wet Wood, 2022; Vyas et al., 2023). In addition, rushing towards area-based targets by certain deadlines – i.e., 2030 - could lead to unsuitable designations that do not align with reconciliation or conservation goals (Zurba et al., 2019). In landscapes outside of Parks Canada's jurisdiction or co-jurisdiction, care will need to be taken to avoid encroaching industrial development through well-designed protected area management and suitable Indigenous involvement (Parlee et al., 2012).

Table 3.4.2.4- *Numbers of parks with each of three effect directions, and whether they are managed with or without Indigenous partners.*

Effect direction	Indigenous co-management	No Indigenous co-management
Positive	2	3
Negative	1	12
Neutral	17	5

Parks Canada's co-management of its protected areas with Indigenous groups has had varying degrees of success. In theory, co-management aims to balance conservation, honouring legal claims to land, granting public access, encouraging self-government and profit (LeBlanc & LeBlanc, 2010).

Of the 20 parks with formal indigenous co-management, most (85%) have had a neutral effect on landscape fragmentation change, in comparison to their respective control areas (Table 3.4.2.4). Only one park-control pair – Tuktut Nogait – has had a very small negative effect direction of around -0.3 km^2 since park designation in 1996. The Tuktut Nogait Management Board has an advisory role, similar to the parks jointly managed by Inuit and Parks Canada: Qausuittuq, Auyuittuq, Quttinirpaaq, Sirmilik, and Ukkusiksalik; and the co-management in Aulavik and Ivvavik with relevant Inuvialuit organizations (Lawson, 1987; Martin, 2016; Bruce & Mulrennan, 2024). Other parks with minimal changes in landscape fragmentation since designation have seen differing relationships between the Crown and local communities. For example, in Nahanni NPR, the Dehcho First Nations have had a successful shared governance arrangement with Canada through the Dehcho Process – self-governance and land/resource planning negotiations, and now through the new Ndahecho Gondi é Gháádé model (Parks Canada, 2022a). Other successful co-management examples include Torngat Mountains NPR and Thaidene Nëné NPR, at which local indigenous stakeholders have been involved in park management since the establishment of the park reserves and Parks Canada has aligned its management goals with the traditional values of the communities (Gibson & Ford, 2023; Parks Canada, 2024). In comparison, the Kluane National Park Management Board demonstrates the evolving nature of co-management, from when hunting and trapping were banned in Kluane's first iteration as a Game Sanctuary in 1943. The indigenous-Crown relationship has improved due to projects that promote Nations' relationships with the land from which they were ostracized half a century prior (Youdelis et al., 2020). A similar evolution is occurring in Vuntut NP, where the park has had a positive effect for effective mesh size (Bruce, 2023). These relationships lay solid foundations for future ecological integrity monitoring and prevention of encroaching development. In contrast, at Gulf Islands NPR and Wapusk NP where there have been negative effect directions for FG2, local First Nations have been generally unsatisfied with Parks Canada's management structures, due to a lack of transparency and improper sharing of information, and the general distrust of federal agencies after longstanding territory disputes and resentment over how the parks were established (Martin, 2006; Bouevitch, 2016). The overarching power imbalance between the Crown and local communities adds complexity to these multilevel co-management cases, despite some decentralization of its administration since the first attempts at Indigenous inclusion in park management (Lawson, 1987; Orozco-Quintero et al., 2020; Bouevitch et al., 2024).

Two parks that have prevented landscape fragmentation in comparison to their control area for certain fragmentation geometries – Pacific Rim NPR (FG2: 0.168 km^2/yr) and Gwaii Haanas NPR (FG1: 0.005 km^2/yr) - are examples of successful co-management. The Pacific Rim NPR falls within the traditional territories of 9 Nuu-chah-nulth Nations, and they form part of cooperative management boards and working groups. Although the NPR was created without the consent or input of these nations, there has been a transition since establishment from a federal dominance over the lands, towards multilevel conservation negotiations and management (Orozco-Quintero et al., 2020). The 2020 park boundaries used in this study include ?A:?b?e:?s a.k.a "Middle Beach" as part of the NPR, which has now been handed back to Pacheedaht First Nation (Baker, 2023), demonstrating Parks Canada's openness to handing back land to the local Indigenous communities. Gulf Islands NPR is cooperatively managed by the Archipelago Management Board (Council of the Haida Nation & Parks Canada, 2018). Foundational to this co-governance is the 'agreement to disagree' on topics such as land ownership and jurisdiction, to focus on shared conservation goals, and the NPR's establishment from the Haida Nation's legal battles to prevent logging and solidify sovereignty over their land (West Coast Environmental Law & Coastal First Nations Great Bear Initiative, 2019).

In contrast, 60% of the park-control pairs with no formal Indigenous co-management plans have had a negative effect direction, and 25% have had little to no effect. The three park-control pairs with a positive effect direction remaining are Elk Island, Pukaskwa, and Grasslands. Overall, parks that are co-managed by Indigenous groups have, so far, been more successful in preventing the expected landscape fragmentation of the region than parks that are not, however it must be noted that these parks tend have been more recently established. As discussed in section 5.3, the more recently established parks have seen less fragmentation change per year than older parks.

3.4.3 Subgroups of the National Park System

3.4.3.1 Fragmentation levels per time-step

The parks designated before 1930, except Wood Buffalo and Elk Island, were mostly created with tourism and recreation in mind, rather than conservation. The railway and related rail-resort parks were mostly the source of anthropogenic fragmentation in the region, bringing with them maintenance infrastructure and resource extraction (Mortimer-Sandilands, 2009). This is consistent with this study's findings that park designation has not been successful in preventing landscape fragmentation and that the rate of decreasing effective mesh size since 1930 has been greater in these parks than their control areas.

The largest effect size (of 11.79 km^2/yr) was for FG2 of parks designated between 1931-1969, including the post-war boom in resource extraction. Park protection was during a time of considerable societal change where designation was key in preventing fragmentation from barriers related to infrastructure development and urbanization/suburbanization (Fairbairn, 1998; McCann, 1999). The results suggest that the parks in this subgroup were successful in preventing the anthropogenic landscape fragmentation over time that occurred in the control areas. The time-step for parks designated between 1970 and 1999 also had a positive effect, likely due to continued suburbanization and an increasing population, despite increases in automobile tourism within the parks (Keddie & Joseph, 1991; Mortimer-Sandilands, 2009). This supports the hypothesis that the longer a national park has been protected, it will have higher landscape connectivity than its associated control area, than pairs that have been protected for less time.

A surprising result from this analysis is that the parks designated during Canada's introduction of ecological integrity monitoring period of 2000-2018 have had a divergence in effective mesh size of over -6 km^2/yr, the largest negative effect size of all the timesteps. This may reflect the prevailing "parks for people" ideology during this time step, although given the remoteness of all the parks designated in this time, except for Gulf Islands, this seems unlikely (Fluker, 2010). It is likely that these parks' short time as protected areas has inflated the rate of change per year, and that over time a continued minimal absolute change in effective mesh size that most park-control pairs in this time step have had would reduce their per-year value.

Since the creation of Canada's first national parks, Canadian tourism has boomed (Statistics Canada, 2017). The oldest parks in the NPS remain the most visited, fueling human development that increases fragmentation and the associated effects on wildlife, keeping a trend of negative divergence of effective mesh size (Rogala et al., 2011). The positive effect of parks designated between 1931 and 1999 demonstrates how introducing conservation into park management has prevented the landscape fragmentation linked to infrastructure development of unprotected areas, creating islands of natural areas.

3.4.3.2 Fragmentation levels per region

The region with the greatest divergence in anthropogenic fragmentation levels over time is the Taiga, with a positive effective mesh size divergence of 25.48 km^2/yr (FG2). The park-control pairs in this region are Wood Buffalo, Nahanni, and Akami-Uapishkᵁ-KakKasuak-Mealy Mountains. The parks in this region have successfully prevented the fragmentation over time that is related to nearby towns and agriculture. The Hudson, Cordilleras, and Prairie park-control pairs showed similar patterns but to a lesser extent. Agriculture is the prevailing barrier fragmenting the control areas in the Hudson and Prairie regions, leading to the positive effect of parks preventing this. A greater rate of effective mesh size change has happened in the unprotected control areas of the Cordilleras region overall than in the parks, also due to agriculture and in addition, unpaved roads related to recreation and industrial development such as mining.

The Pacific Maritime parks of Pacific Rim, Gwaii Haanas, and Gulf Islands have had the largest negative divergence in effective mesh size in comparison to their control areas. These parks have had a greater rate of change of landscape fragmentation than their respective control areas (a divergence of over -12 km^2/yr), despite a history of commercial logging prior to their designation and logging that has continued on in their control areas (Parfitt, 2019). Pacific Rim and Gulf Islands NPRs and their buffer zones are popular tourist destinations and have seen an increase in both tourist and residential developments since their designation, whereas logging-related fragmentation has not increased at the same rate (Frater & Valour, 2020).

As per the Statistics Canada Index of Remoteness (2023), the most remote region in this analysis is the Arctic. The individual parks in this subgroup each have a small effect size, but together the region has an average effective mesh size divergence of −4.3 km^2/yr. This follows the trend for parks designated in the 2000-2018 time-step, a subgroup containing mostly remote parks, that has an inflated per-year effect size. There is a lack of human development in the parks and control areas of the Arctic, but much of the accounted for fragmentation relates to tourism and research activities in the northern parks. The least remote region is the Atlantic Maritime & Great Lakes, which also has a negative effect size, but not to the same extent, of under -1 km^2 /yr. As the population is high in this region and transport infrastructure is well connected, travel tourism is high in this region, thus explaining the negative effect direction for this region. The Cape Breton Highlands, Prince Edward Island and Fundy park-control pairs would likely have changed this figure if they had been able to be included in this analysis as they make up 3 of the 7 parks in the Atlantic Maritime & Great Lakes.

3.4.4 Limitations

Although the data used in this analysis cover most of Canada temporally and spatially, there are some park-control pairs were missing time-steps of data. For example, data availability in the Atlantic provinces was lower than in the west before 1950, and so statistical analysis had to be skipped for the park-control pairs of Fundy, Prince Edward Island, and Cape Breton Highlands. When pairing each park with an appropriate control area, it was not possible to locate a suitable control area for a few island parks – Georgian Bay Islands, Mingan Archipelago, Sable Island, and Thousand Islands – due to their unique land cover, shorelines, and area extent. This affected the completeness of the study on the NPS and overlooked potential insights into human impacts on the fragile island ecosystems.

Regarding the topographical features of the data: some forestry and resource roads may have been removed from datasets over time and new ones are not included in the datasets due to their status on private land, thus falsely inflating the effective mesh size in these areas & underestimating the level of fragmentation (CanVec Series, 2021; Government of British Columbia, 2024). Other fragmenting barriers/connecting elements that would have been included in this study had they been available across each time-step in the data are underpasses and bridges, fences, powerlines, parking lots and seawalls (Bartzke et al., 2014; Jakes et al., 2018). In addition, the data across the study's timeline do not include indirect anthropogenic causes of landscape fragmentation such as fire-related impacts and melting ice from climate change (Driscoll et al., 2021; Kellner et al., 2023).

Although a single Before sample is generally insufficient for model development in BACIPS designs (Osenberg et al., 2011), it is somewhat accounted for in the PC BACIPS model that focuses on the trend of data After an impact (Thiault et al., 2017). In addition, temporal pseudoreplication can become an issue when measurements are taken closer together in time (Manly, 2009): particularly for newer parks like Thaidene Nëné. This stems from the limitations of the data availability, and the likely results from pseudoreplication is that effects appear to be more statistically significant than they should be.

3.4.5 Looking to the future: landscape fragmentation in national parks and their greater park ecosystems.

The main goals shaping Canada's action towards preventing landscape fragmentation are the Pathway to Canada Target 1 and 30x30. Federal targets from both Parks Canada and Environment and Climate Change Canada regarding landscape fragmentation appear to be stemming directly from these goals (Parks Canada, 2023d). For example, the National Program for Ecological Corridors follows goals in Pathway to Canada Target 1 described as identifying and documenting areas that are not currently protected but are important for connectivity, and thus particularly important for preventing fragmentation (P2C T1, 2021). This program acknowledges the lack of connectivity of Canadian National Parks with their surrounding landscape and is creating tools and resources to enable independent corridor initiatives such as the Cootes to Escarpment EcoPark Ecological Corridor Pilot Program and the Consolidation of the Forillon Ecological Corridor (Beazley & Hum, 2021; Parks Canada, 2023f). The program has identified 23 corridor priority areas in the country that are outside the scope of Parks Canada's administration. Here, support to and collaboration with local conservation initiatives would be a good management strategy.

However, the National Program for Ecological Corridors does not cover how Parks Canada can directly improve on landscape fragmentation monitoring or prevention. Parks Canada's strategy for contributing to the 30x30 goal is to complete the NPS by representing all terrestrial ecoregions. Since the finalization of this study, a new NPR was designated: Pituamkek (Hog Island Sandhills) in PEI. Further to this, there are proposals for new parks in South-Okanagan-Similkameen, BC and the Seal River Watershed, Manitoba. These new protected areas will contribute to the 30x30 goal by area. However, it is not yet clear how landscape fragmenting elements in and around these parks will be managed due to undecided boundaries and lease disputes (Parks Canada, 2022b). Parks Canada is looking to prevent encroaching human development on natural areas near to population centres by focusing on new National Urban Parks, similar to Rouge Urban Park in Toronto. Potential urban areas currently in discussion for an Urban Park designation include Edmonton, Windsor (Ojibway Prairie Complex) and Halifax (Blue Mountain-Birch Cove Lakes) (Parks Canada, 2021a). The agency is also working on welcoming more Canadians and international tourists to its park system, with the goal of "creating a culture of stewardship" amongst visitors (Parks Canada, 2017b:4). The busiest parks with the highest fragmentation, such as Banff and Waterton Lakes, will need to be managed and invested in to provide improved infrastructure that does not impede wildlife movement.

With these new parks and increasing tourism in mind, will ecological integrity monitoring (EIM) continue to be a management strategy for Parks Canada, and will landscape fragmentation be effectively monitored within it? Evaluations of the program show that there has been significant work to implement EIM policies and guidelines, but systems for monitoring and reporting on indicators have been very slow and a decrease in funding for the agency undermines lofty goals of improvement (OAGC, 2013; Parks Canada, 2023b). Although there have been efforts to incorporate landscape-scale monitoring into EIM, there are still opportunities for further development and inclusion of fragmentation indicators. It has also been noted that EIM should include indicators to better monitor indirect causes of fragmentation, such as climate change and fire patterns (Parks Canada, 2023b). This study gives an overview of fragmentation in the NPS over most of its existence and compares it to similar unprotected areas. It assessed the impact of landscape fragmentation on individual national parks and the system as a whole; setting a solid foundation for future, species-agnostic landscape-scale monitoring of barriers affecting animal movement.

3.5 - Conclusions and future research suggestions

Landscape fragmentation is a key underlying aspect of ecological integrity that Parks Canada includes in its EIM framework (DeFries et al., 2007; Warner, 2008). Fragmenting elements can result from human impacts and natural processes, however this study has found that human influence on Canadian landscapes is the more considerable driver of fragmentation. Both in and around parks, disruption to natural land cover caused by fragmentation threatens biodiversity, ecosystem services and perceived landscape value (Jaeger et al., 2007; Wartmann et al., 2021). This study analysed the divergence of landscape fragmentation over 90 years between Canadian national parks and control areas, employing the Effective Mesh Size (m_{eff}) metric and a PC-BACIPS study design.

Overall, the findings indicate that park protection across the NPS has been relatively successful in preventing landscape fragmentation compared to unprotected control areas. However, older parks with longer history of human influence on the landscape are more fragmented than their control areas, and fragmentation levels have gradually worsened with time. Meanwhile, parks that have been designated more recently (in the last 50 years) tend to have a greater effective mesh size than their control areas, with the control areas increasing fragmentation levels faster than in the parks over time. Some regions have had more success than others in preventing a reduction of Effective Mesh Size, with parks in the Taiga and Hudson having the greatest protective effect, but Pacific Maritime parks becoming more fragmented than their control areas over time. As expected, the more remote parks in the NPS have the largest effective mesh sizes, due to their size and a lack of anthropogenic influences in and around their boundaries. However, their rate of change since designation is greater than in other regions and in a negative direction, indicating that fragmentation levels in remote parks are increasing faster than in less remote regions. According to the PC-BACIPS test, for the NPS and for the time-step and regional subgroups, a linear model was the most suitable for assessing the data based on its trends after park designations.

These findings can help inform ecological connectivity programming and management, especially where national parks and park reserves are key anchors in large landscape ecological corridors (Staying Connected Initiative, 2023; Yellowstone to Yukon Conservation Initiative, 2024). The large study size covering the majority of the NPS and the long time period studied provide a high-level analysis of fragmentation in Canadian national parks, enabling effective comparisons between individual parks, regions, and park ages. The results highlight regions and individual parks needing the most attention to reduce human fragmentating influences, and identify those requiring less intervention within and adjacent to their boundaries.

This study also offers valuable insights into the historical trends of fragmentation in the NPS, the main drivers of this fragmentation by considering different fragmentation geometries, and the impact of different management strategies over time, demonstrating that when the "parks for people" ideology has prevailed in park management, there has been an increase in anthropogenic fragmentation in the parks (Fluker, 2010). Despite a trend towards conservation-focused park management, these results show that there is a need for Parks Canada to prevent further landscape fragmentation and include more landscape fragmentation-related monitoring into their EIM framework. Effective park management with landscape fragmentation and connectivity at its forefront will help Canada meet its goals relating to the Pathway to Canada Target One initiative and Global Biodiversity Framework's 30x30 target (WWF & IUCN, 2023).

Future studies about the fragmentation in provincial protected-area systems using the same or similar methods to those employed here would provide interesting insights into how different agencies approach park management in relation to landscape fragmentation. This would also offer a more thorough analysis of priority regions for ecological connectivity, and the provincial parks' resources and abilities to provide large, connected areas of natural habitat for wildlife.

The data used in this study do not contain all private roads, resource extraction impacts, or powerlines, nor habitat type or quality in the landscape patches, due to the lack of consistency and availability in maps across all time-steps. Remote sensing technologies and an analysis of more recent time-steps could be used to understand how these factors have influenced effective mesh size in and around Canadian national parks.

4 - Overall conclusion

Actions are required to reduce the impacts of landscape fragmentation, reduce any further fragmentation, and to improve the functional connectivity in and around Canadian national parks. An over-arching theme of heavy human impacts on parks and greater park ecosystems, despite conservation-focused policies, demonstrates that it is essential to identify trends of changes in ecological indicators to enable park management to comply with legislation. Landscape fragmentation is connected with almost all of the ecological stressors on Canadian national parks (DeFries et al., 2007; Warner, 2008). Monitoring and reporting on landscape fragmentation would contribute to effective park management in order for Canada can meet its goals relating to the Pathway to Canada Target One Initiative and the Global Biodiversity Framework's 30x30 target (WWF & IUCN, 2023).

This study analysed the divergence of landscape fragmentation over 90 years between Canadian national parks and paired control areas, employing the effective mesh size (m_{eff}) metric and a PC-BACIPS study design. Up to five fragmentation geometries were considered for the NPS, each subgroup, and each park-control pair, to account for the different reactions of species groups to certain barriers. Fragmentation can result from both human impacts and natural processes. This study included multiple fragmentation geometries to analyze both, but mainly focused on changing management approaches and anthropogenic fragmentation. The results show that park protection across the NPS has been somewhat successful in preventing landscape fragmentation in comparison to the unprotected control areas. In general, parks and control areas were more fragmented after the date of park designation than before. Some regions have had more success than others in preventing a reduction of effective mesh size, with parks in the Taiga and Hudson having the greatest protective effect, but Pacific Maritime parks becoming more fragmented than their control areas over time. As expected, the more remote parks in the NPS have the largest effective mesh sizes due to their size and a lack of anthropogenic influences in and around their boundaries. However, their rate of change since designation is larger than in other regions and is in a negative direction, indicating that fragmentation levels in remote parks are increasing faster than in less remote regions. The older parks, with a long history of human influence on the landscape, tend to be more fragmented than their control areas, with levels of fragmentation gradually worsening over time after 1930.

According to the PC-BACIPS test, for the NPS as a whole and for the time-step and regional subgroups, a linear model was the most suitable for assessing the data. For the individual park-control pairs, a linear model was selected for the majority with occasional step- and sigmoid- models selected. Only two park-control pairs had the asymptotic model selected for some of their fragmentation geometries.

This study's findings can help inform ecological connectivity programming and related management, especially where national parks and park reserves are considered the cornerstones of important large-scale ecological corridors (Pither et al., 2023; Yellowstone to Yukon Conservation Initiative, 2024). The regions and parks that require the most focus in reducing anthropogenic influences within and adjacent to their boundaries are highlighted, as are the most common types of fragmenting elements affecting wildlife movement in the parks and control areas.

The results also offer insights into the historical trends of fragmentation in the NPS, which can be related to the impacts of different management strategies over time. Despite legislation requiring ecological integrity be maintained or restored in Canada's national parks, the increasing fragmentation of many parks in the NPS shows that in the balance of conservation and tourism, the "parks for people" ideology has been dominant. Parks Canada should include more species-agnostic landscape fragmentation-related monitoring and reporting into their EIM framework, in order to combat this. Now that there are effective mesh size values available at time-steps before designation for the majority of parks in the Canadian NPS, this study design should be used at future time-steps to extend this analysis and conduct ongoing landscape fragmentation monitoring. It should also be used for any new parks in the NPS, because data for the "before" time-step can be easily extracted using CanVec.

The nuances associated with the movement and connectivity of particular species and their habitats across the NPS and in each greater park ecosystem, could not be fully considered in the large-landscape scale of this research. However, the strength of this analysis stems from its wide geographic range, long temporal range, and consideration of multiple species types through the fragmentation geometries. In addition, a single Before sample is generally insufficient for model development in BACI designs (Osenberg et al., 2011), but this was somewhat accounted for in the PC BACIPS design used here by considering the trend of data After park designation

(Thiault et al., 2017). Due to data availability, some individual park-control pair results may appear more statistically significant than they should be, due to temporal pseudoreplication. This is more likely for the more recently designated parks, where measurements of effective mesh size were taken closer together in time than for in the older parks (Manly, 2008). The difficulty of finding suitable and consistent map data for this study meant that some potentially useful landscape elements were not included, such as parking lot and powerline features, or unofficial and private roads and trails that would also be affecting wildlife (Spernbauer et al., 2023). These could cause particular issues in control areas where off-trail recreation or resource extraction is not as closely monitored as in protected areas. Remote sensing technologies could be used for recent time-steps to understand how these factors have influenced fragmentation in and around Canadian national parks and their control areas.

The same, or similar, methods to those used in this study could be used in future studies to evaluate how other agencies approach park management in relation to landscape fragmentation, such as provincial protected area networks, other national park systems, or cross-boundary connectivity initiatives. This would offer a more complete analysis of priority regions for ecological connectivity in Canada and elsewhere, and would provide insight into other systems' resources and abilities to maintain large, connected natural landscapes for wildlife connectivity. For other park systems with similar settler-colonial backgrounds, such as the United States and Australia, this study design could facilitate a comparison with the fragmentation history of Canada's NPS. However, a strength of this study of federally protected areas in Canada is that there are enough data with suitable spatiotemporal resolution to cover the long study time period of more than 90 years. This may not be the case when studying other park management systems elsewhere. Further uses of this study design within the jurisdiction of Parks Canada can also include National Historic Sites, National Urban Parks, and National Marine Conservation Areas.

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6 – Appendices

6.1 Control area suitability table

Name	Suitable control area location	Area extent	Proximity to park, at closest point	Ecoregion/ecoprovince	Waterbody comparison (% of park area)	Overall land use/cover	Rating (out of 8) Control Area similarity index
		1 - same as park	1 - under 100km 0 - over 100km	2 - all same 1 - one the same 0 - different	2 - under 5% 1-under 10% 0 - over 10%	1 - some similar. 2 - totally similar	
Banff	South west, into BC. south end by Kimberly and Cranbrook	Same as park	61km	Columbia Montane Cordillera	1.28%	Rangeland and pasture, mixed forest, perennial snow and ice, barren land	8
Glacier	Directly north west	Same as park	3.7km	Columbia Montane Cordillera	0.013	Rangeland and pasture, mixed forest, perennial snow and ice, barren land	8
Yoho	North west, adjacent to the north end of Banff	Same as park	28km	Columbia Montane Cordillera	3.28%	Rangeland and pasture, mixed forest, perennial snow and ice, barren land	8
Waterton Lakes	Directly north, rotated about 30 degrees to include similar mountains and water	Same as park	2.3km	Columbia Montane Cordillera/Central Grassland (control area)	5.83%	Barren land, rangeland and pasture, mixed forest, cropland (not in park)	5
Jasper	Slight north west into BC		2km	Columbia Montane Cordillera	0.25%	Rangeland and pasture, mixed forest, perennial snow and ice, barren land	8
Elk Island	Directly south east	Same as park	8.7km	Parkland Prairies	3.27%	Broadleaf Forest, Rangeland and pasture, cropland, mixed forest	8
Mount Revelstoke	South of highway, rotated to include icefields	Same as park	0.1km	Columbia Montane Cordillera	0.02%	Rangeland and pasture, mixed forest, perennial snow and ice, barren land	8
Point Pelee	North east along coastline. Island control on Pelee island	Same as park	1km	Huron-Erie Plains	9%	Built up area, mixed forest, barren land, cropland, broadleaf forest	7

Kootenay	South of Banff, near Fernie	Same as park	58.5km	Columbia Montane Cordillera	0.27%	Rangeland and pasture, mixed forest, perennial snow and ice, barren land	8
Wood Buffalo	Slightly north west	Same as park	3.8km	Hay-Slave Lowlands/Central Boreal Plains	4.36%	Forest, cropland, rangeland and pasture	8
Prince Albert	Directly west	Same as park	2.8km	Central Boreal Plains	1.47%	Broadleaf Forest, Rangeland and pasture, cropland, mixed forest	8
Riding Mountain	Directly south & rotated 180 degrees	Same as park	0.6km	Central Boreal Plains/Parkland Prairies (control area)	0.15%	Coniferous forest/mixed forest/cropland/rangeland and pasture	6
Cape Breton Highlands	South & rotated 180 degrees	Same as park	5.4km	Fundy Uplands	0.98%	Sparsely Vegetated/Barren land, Mixed Forest, cropland, built up area	8
Prince Edward Island	East and west along coastline	Same as park	0.3km	Northumberland Lowlands	5.86%	Mixed forest, cropland, barren land	7
Fundy	Directly west	Same as park	36km	Fundy Uplands	0.65%	Broadleaf Forest, Built Up Area, mixed forest, barren land, cropland	8
Terra Nova	South-east	Control - different shoreline	4.7km	Newfoundland	2.70%	Mixed forest, transitional forest, barren land	7
Kejimkujik	North east, central peak similar. Seaside on similar coastline	Control - different shoreline	11.1km	Fundy Uplands	2.86%	Broadleaf Forest, Built Up Area, mixed forest, barren land, cropland	7
Kouchibouguac	South, by sand dunes	Control has more waterbodie s	4.1km	Northumberland Lowlands	1.64%	Mixed forest, cropland, barren land	7
Forillon	Headland directly south	Same as park	6.9km	Appalachian-Acadian Highlands	0.26%	Coniferous forest/mixed forest/cropland	8
La Mauricie	South west, avoiding provincial protected areas	Same as park	15.8km	Southern Boreal Shield (both), Great Lakes-St.Lawrence Lowlands (control)	2.97%	Mixed forest, cropland, built up area (control)	6
Pacific Rim	Removed broken islands group from analysis - cannot find	Control - different shoreline	0.45km	Southern Coastal Mountains, Georgia Depression (control)	0.89%	Barren land, mixed forest, perennial snow or ice	5

	control area. West coast trail unit south down the coast. Long beach unit control on Nootka island.					(park), cropland & built up area (control)	
Auyuittuq	West + rotated to include glaciers	Same as park	206km	Southern Arctic Cordillera (park), Baffin Uplands (Control)	0.89%	Barren land, tundra, perennial snow and ice	5
Kluane	South east & rotated. NP and NPR split	Same as park	394km	North coastal mountains, Wrangel mountains, Southern Boreal Cordillera, Northern Montane Cordillera	0.02%	Tunda, sparsely vegetated land, mixed forest	6
Nahanni	North in same mountain range, rotated slightly	Same as park	5km	Mackenzie-Selwyn Mountains, Mackenzie Foothills, Northern Boreal Cordillera	0.34%	Barren land, Mixed forest, Tundra	7
Gros Morne	Further up the coast by Port Saunders	Same as park	14km	Newfoundland	5.75%	Barren land, coniferous forest, mixed forest	7
Pukaskwa	West along coastline	Same as park	65km	Mid-Boreal Shield	3.20%	Mixed forest	8
Grasslands	Same latitude east and west, western most block is north	Same as park	1.5km	Central Grassland	0.38%	Both: mixed forest, rangeland and pasture, cropland	8
Ivvavik	East on coastline	Same as park	1km	Old Crow-Eagle Plains, Northern Yukon Mountains, Amundsen Lowlands	0.38%	Mixed forest, Tundra, Barren land	8
Bruce Peninsula	South-east, further back into the peninsula	Same as park	17.5km	Great Lakes-St.Lawrence Lowlands	3.03%	Both: Built up area, coniferous forest, mixed forest, broadleaf forest	8
Gwaii Haanas	North of the islands on Graham Island. Each smaller parcel boundary moved around to solve coastline problem	Same as park	10.5km	Southern Coastal Mountains	4.93%	Both: Perrenial snow or ice, Coniferous forest, mixed forest, sparsely vegetated	8

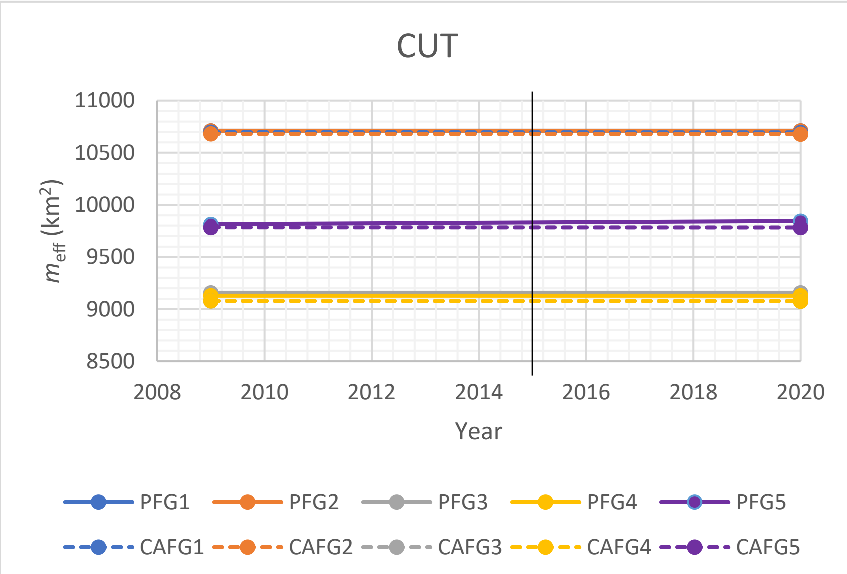
Quttinirpaaq	Not sure about boundary	Control - different shoreline	44km	Northern Arctic Cordillera, Ellesmere Basin	0.04%	Perennial snow or ice;Tundra; Sparsely Vegetated/Barren	7
Aulavik	South-east on Banks Island, along south coast and rotated 140ish degrees CHECK	Same as park	158km	Victoria Lowlands	3.85%	Barren land, Tundra	7
Vuntut	Directly south of Ivvavik control area, like the parks	Same as park	0.2km	Old Crow-Eagle Plains, does not include Northern Yukon Mountains	5.20%	Mixed forest, Tundra, Transitional Forest	5
Wapusk	Rotated to run along coastline south-east	Same as park	7.5km	Hudson Bay Coastal Plains, Hudson James Lowlands	11%	Tundra, transitional forest (only one similar), continuous forest, mixed forest	5
Tuktut Nogait	East in Nunavut, avoinding Inuvialut lands	Same as park	56km	Amundsen Lowlands	5.77%	Tundra, transitional forest, barren land	7
Sirmilik	4 different: three south-west of original, bylot island CA south-east on coast in arctic cordillera	Same as park	7km	Parry Channel Plateau, Boothi-Foxe Shield, Southern Arctic Cordillera, Baffin Uplands (small amount)	5.53%	Perennial snow or ice;Tundra; Sparsely Vegetated/Barren	7
Gulf Islands	Different, unprotected islands with similar land cover used	Same as park	0.02km	Georgia Depression	2.06%	Mixed Forest, Coniferous Forest, Broadleaf Forest, Sparsely vegetated/barren land	8
Ukkusiksalik	South by Baker Lake, twisted around 15 degrees clockwise	Same as park	110km	Mostly in Boothia-Foxe shield but a very small bit of control area in Keewatin Lowlands due to proximity to lake coastline	1.80%	Sparsely Vegetated/Barren land. Tundra: treeless arctic and alpine vegetation. in both	7
Torngat Mountains	South-east down the coastline, twisted clockwise	Same as park	0.5km	Southern Arctic Cordillera/Labrador Uplands	4.80%	Tundra and Sparsely vegetaded/Barren land	6
Nááts'įhch'oh	North	Same as park	12km	Mackenzie-Selwyn Mountains	0.25%	Sparsely Vegetated/Barren land. Tundra: treeless arctic	8

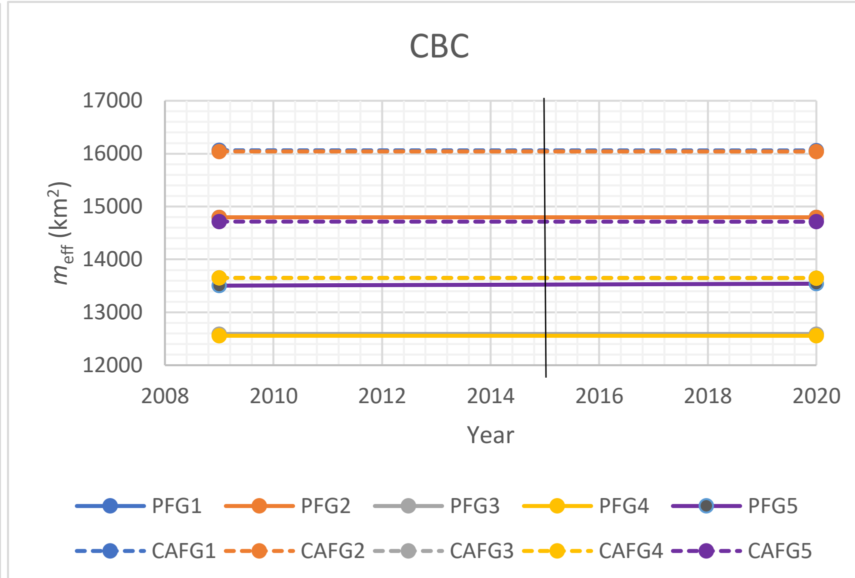
						and alpine vegetation. Coniferous Forest	
Akami-Uapishkᵁ-KakKasuak-Mealy Mountains-	Directly west, next to Lake Melville and covering Happy Valley-Goose Bay	Same as park	45km	Eastern Boreal Shield and Labrador Uplands in both	0.34%	Mixed Forest, Coniferous Forest, Transitional Forest, Sparsely vegetated/barren land	7
Qausuittuq	East to Devon Island. Islands placed near coastline	Same as park	185km	Victoria Lowlands and Parry Channel Plateau - similar elevation, under 500m	5.20%	Sparsely Vegetated/Barren land. Tundra: treeless arctic and alpine vegetation	4
Thaidene Nëné	Rotated to run along coastline to the north west of park.	Same as park	5km	Western Taiga Shield	2%	Tundra/transitional forest	8

6.2 Effective mesh size results for individual park-control pairs

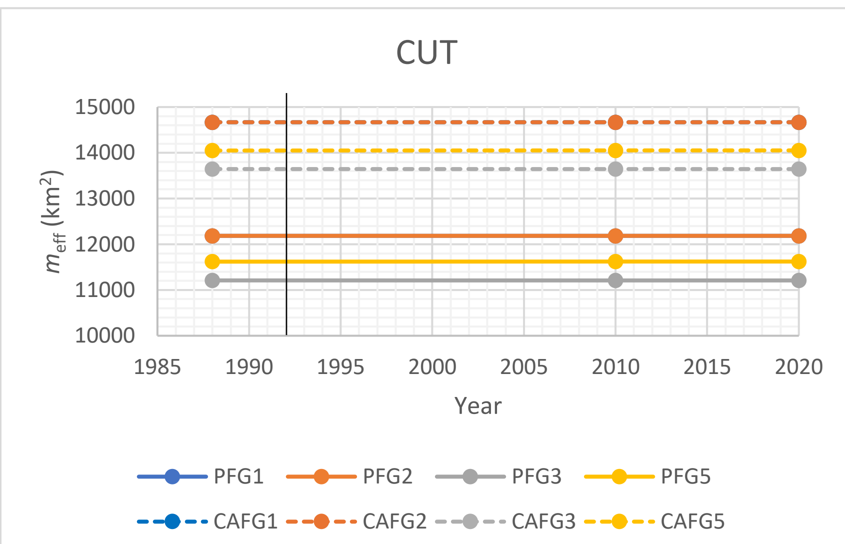
Parks designated before 1930 were considered "conserved" after the National Parks Act of 1930.

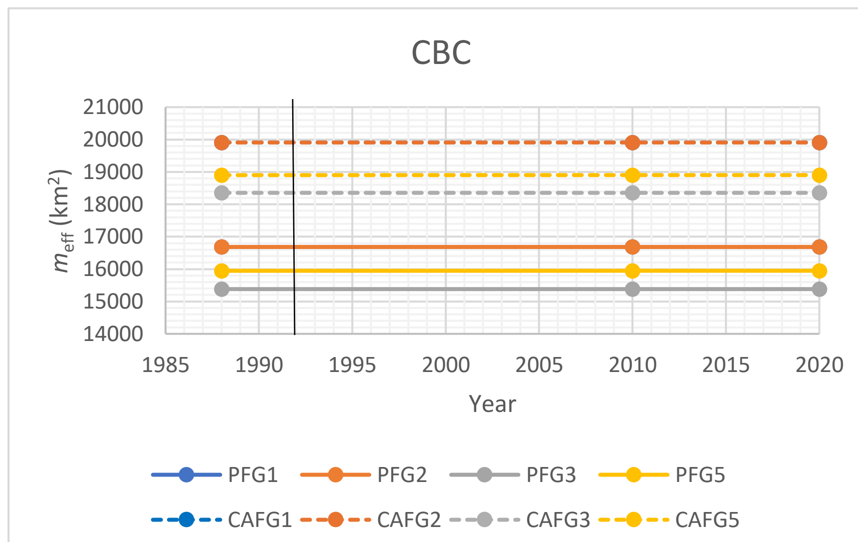
Akami-Uapishk^{U}-KakKasuak-Mealy Mountains, designated 2015



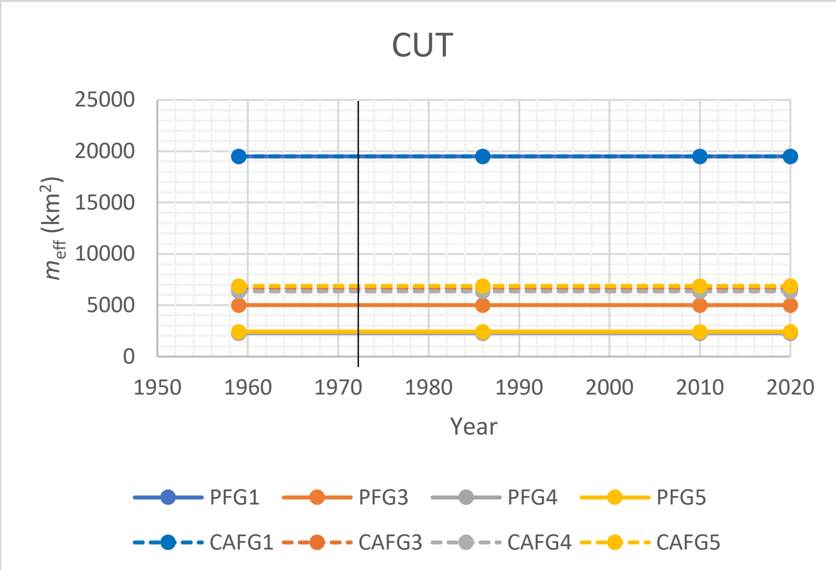


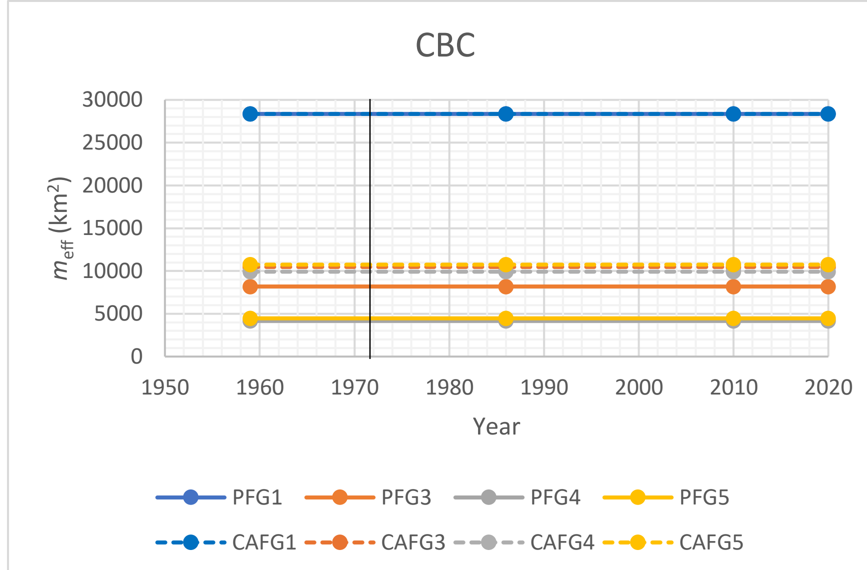
Aulavik, designated 1992



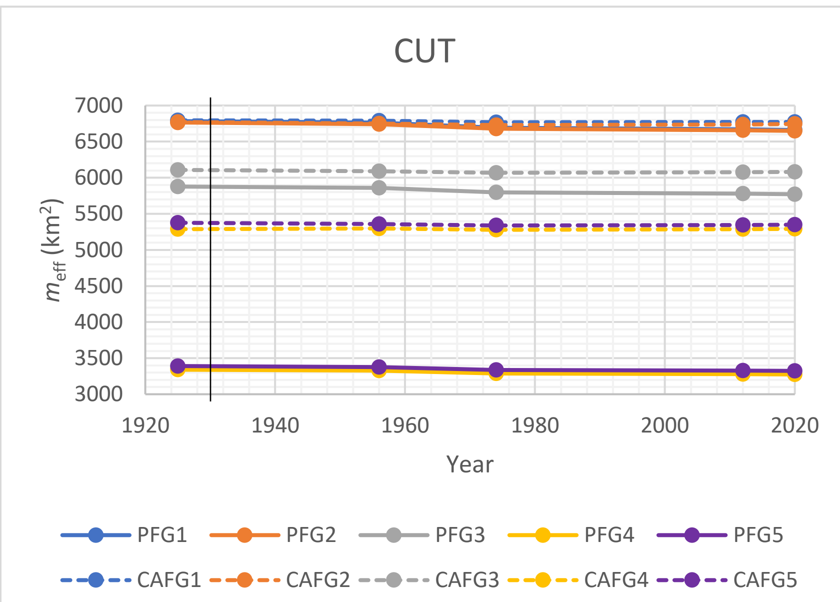


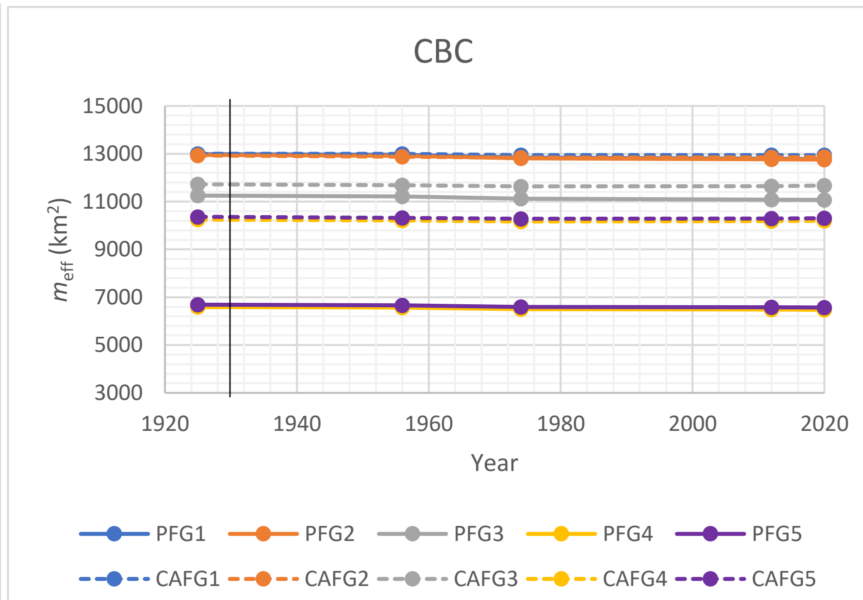
Auyuittuq, designated 1972



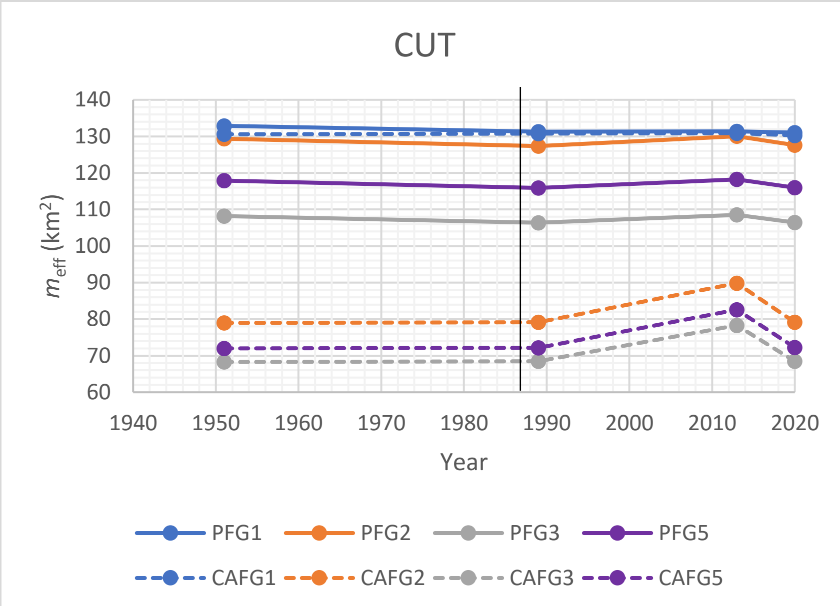


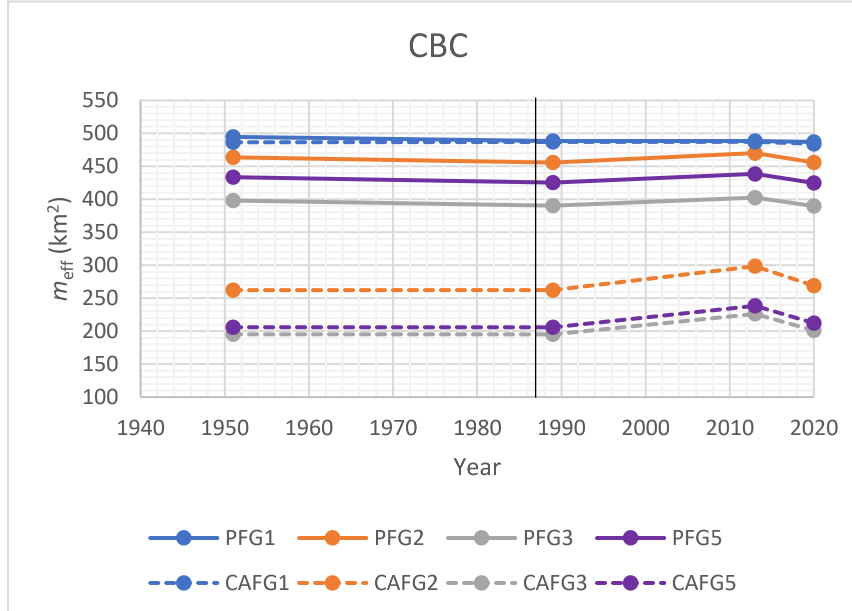
Banff, designated 1882



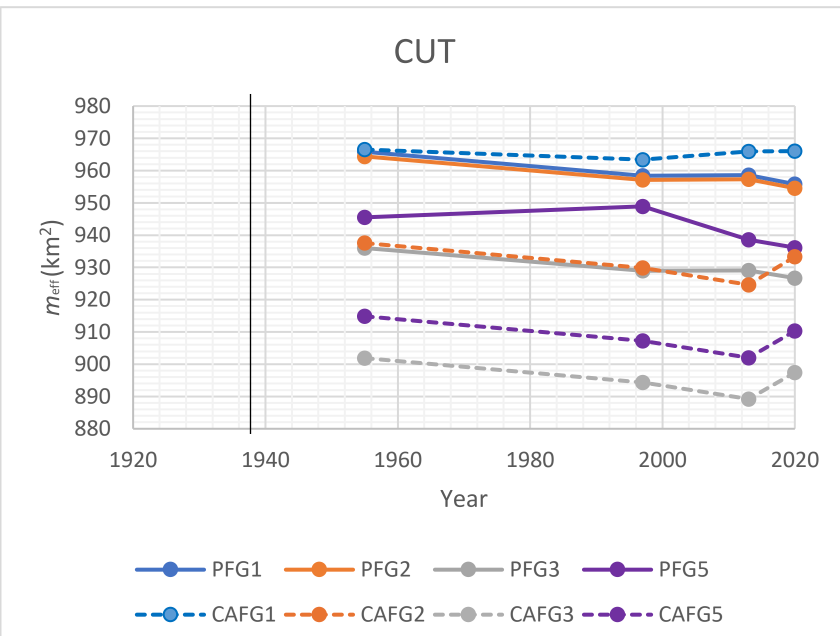


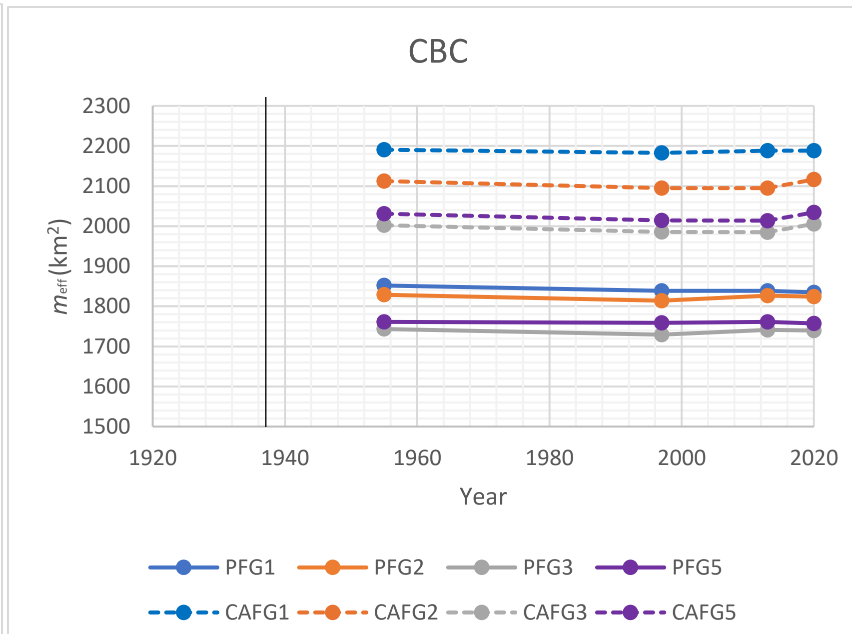
Bruce Peninsula, designated 1987



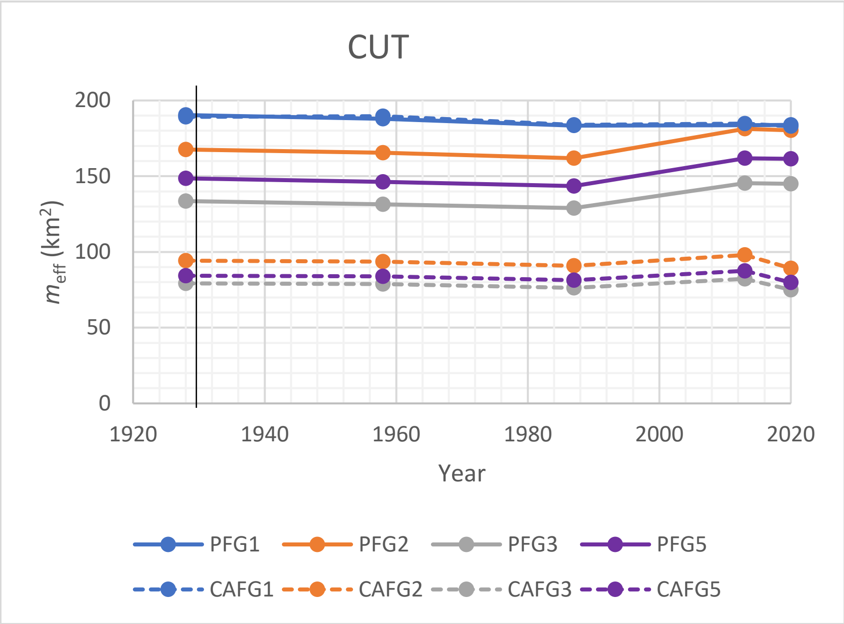


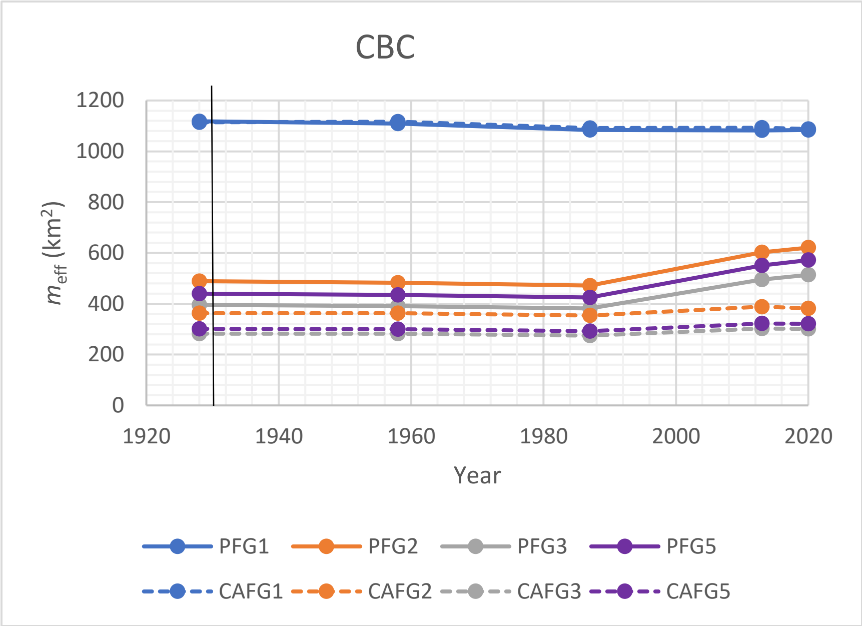
Cape Breton Highlands, designated 1936



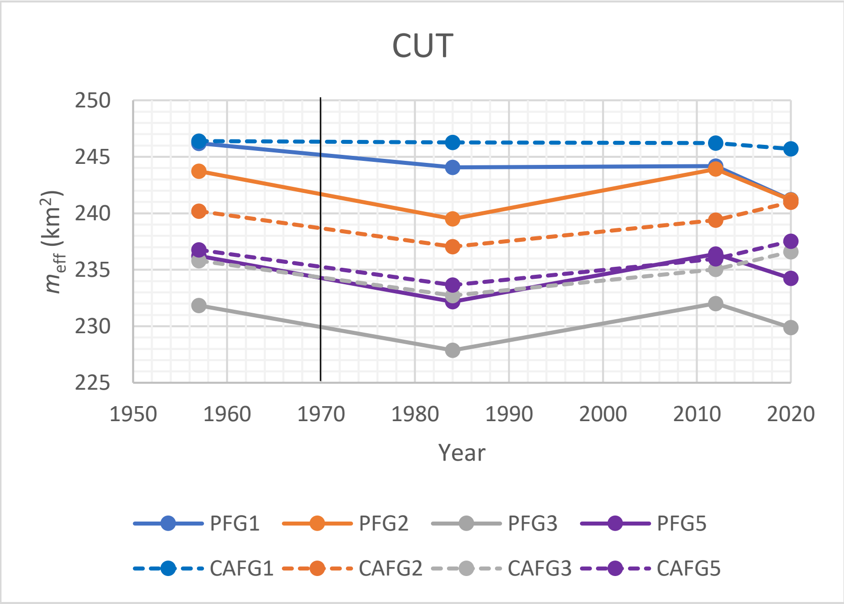


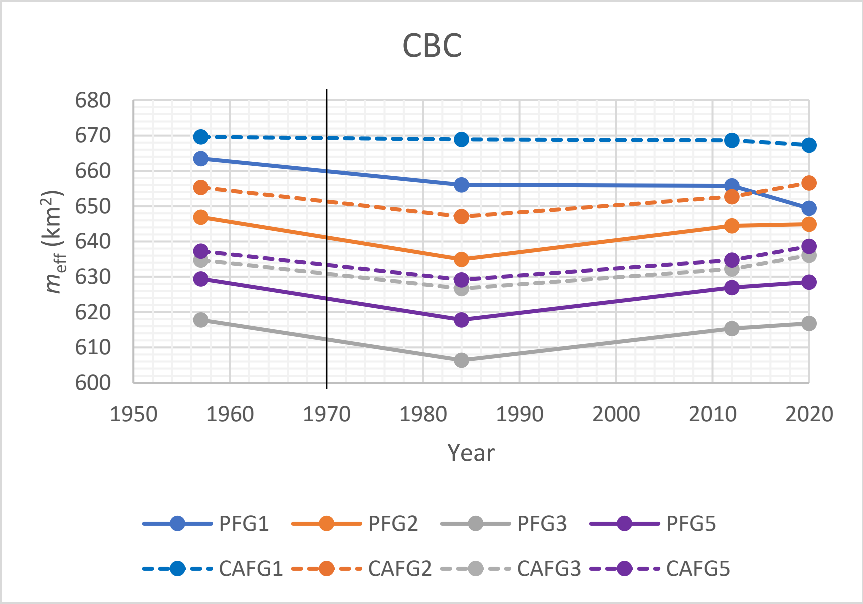
Elk Island, designated 1913



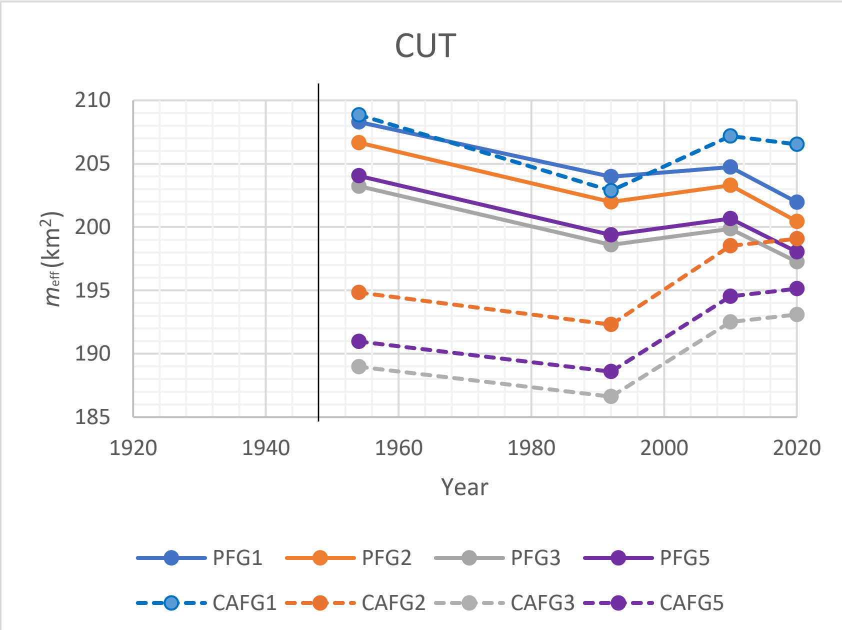


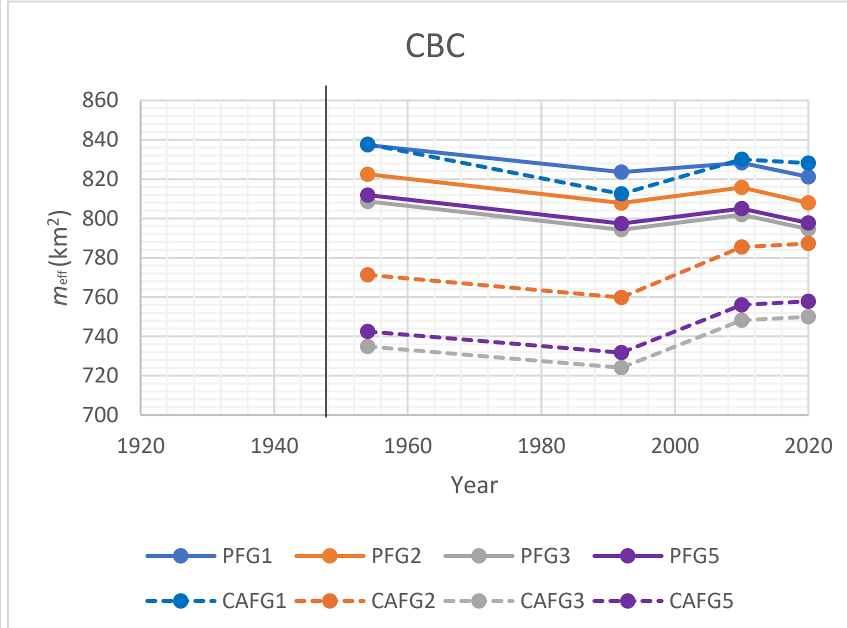
Forillon, designated 1970



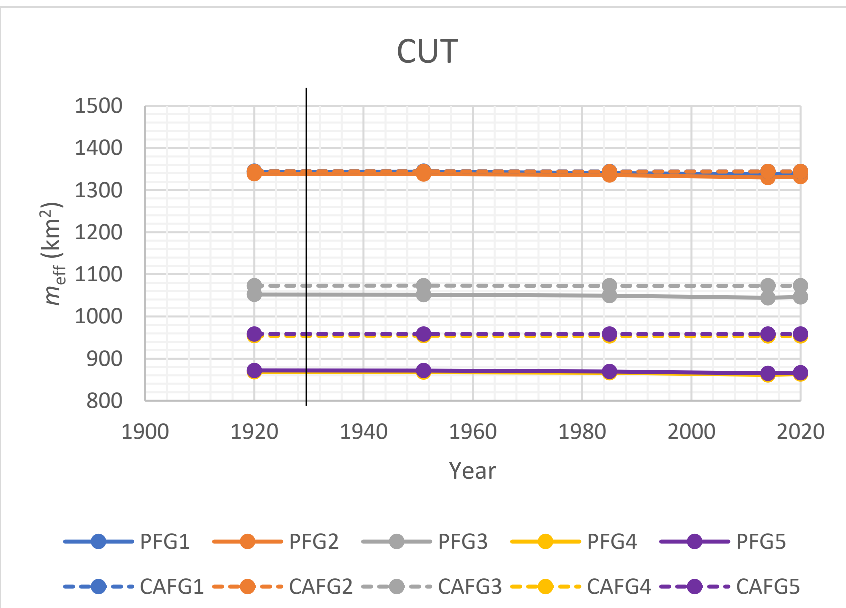


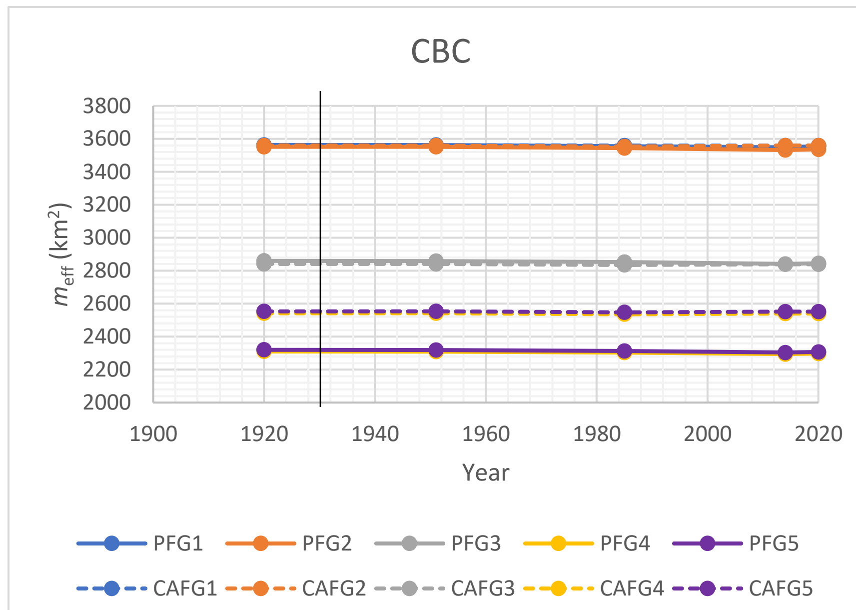
Fundy, designated 1948



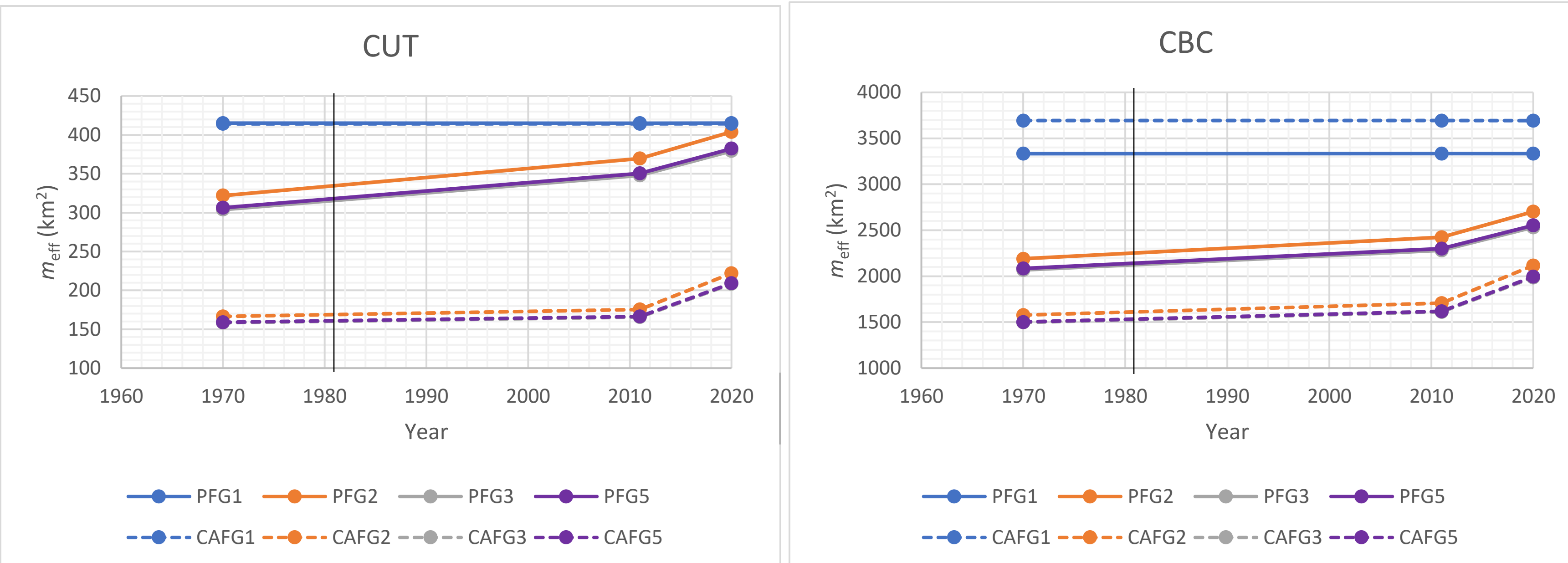


Glacier, designated 1886

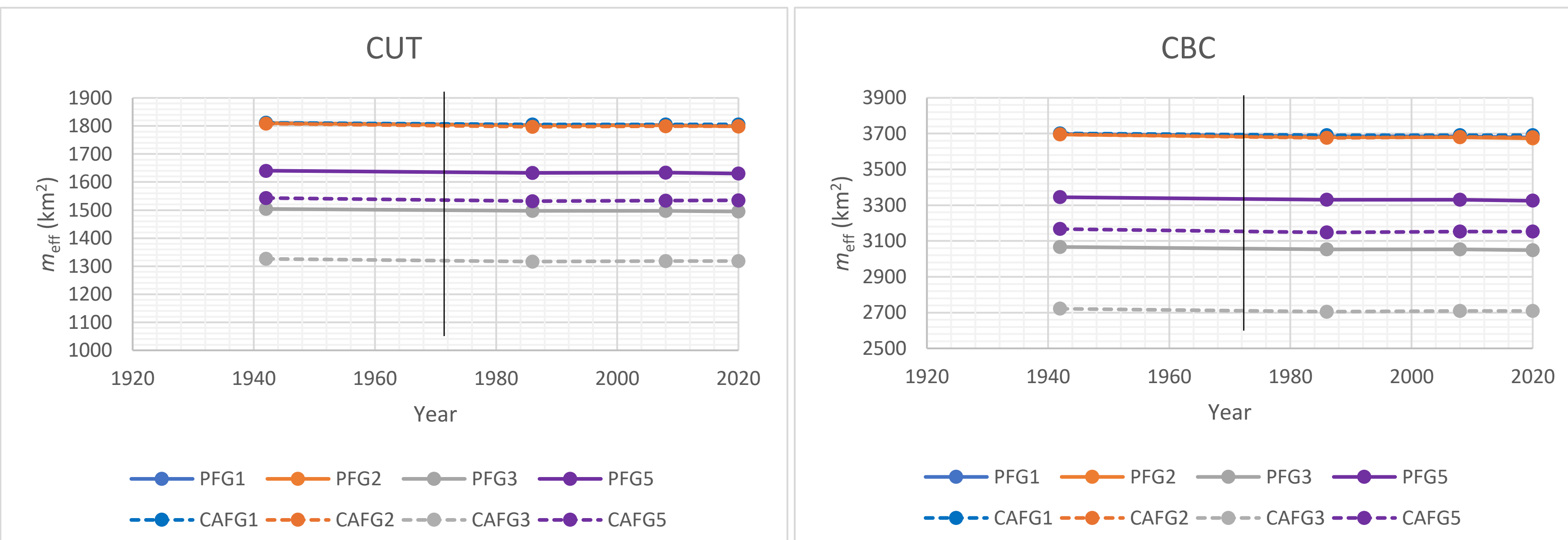




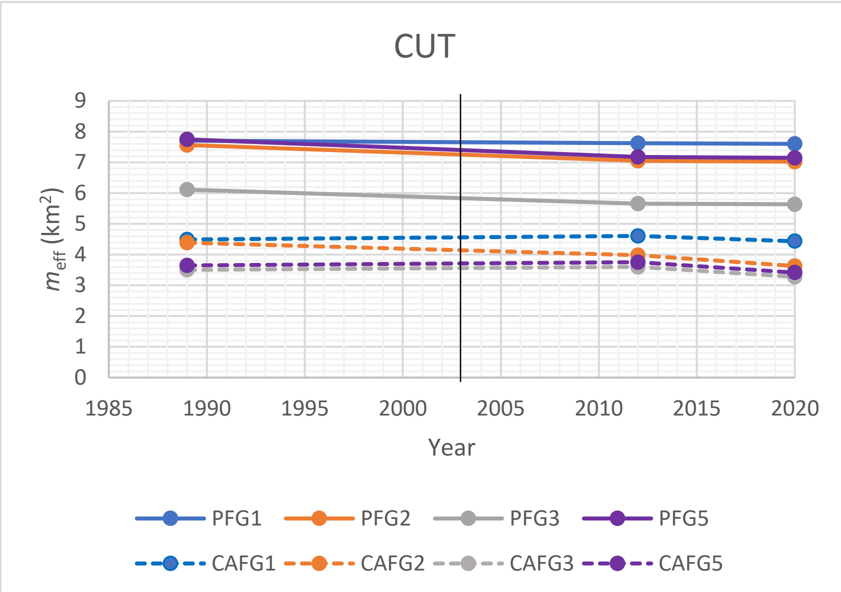
Grasslands, designated 1981

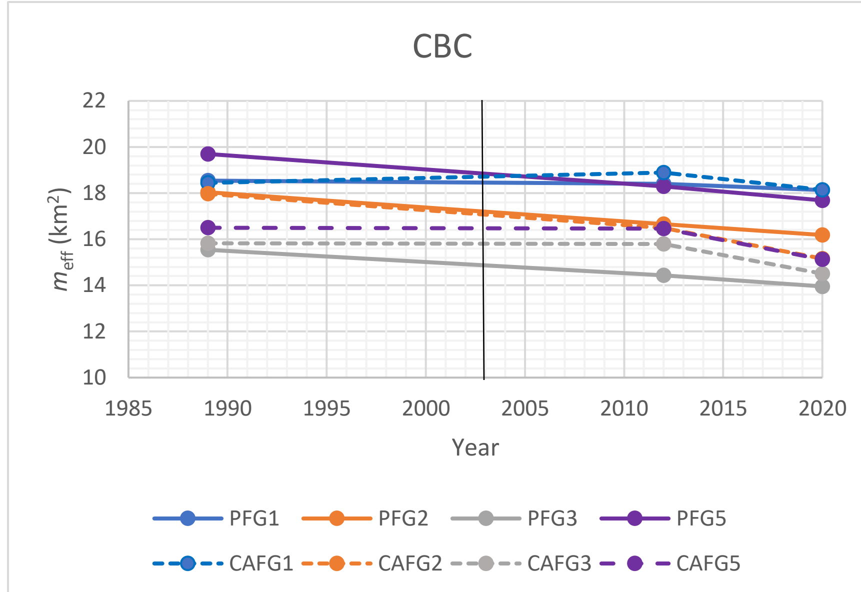


Gros Morne, designated 1973

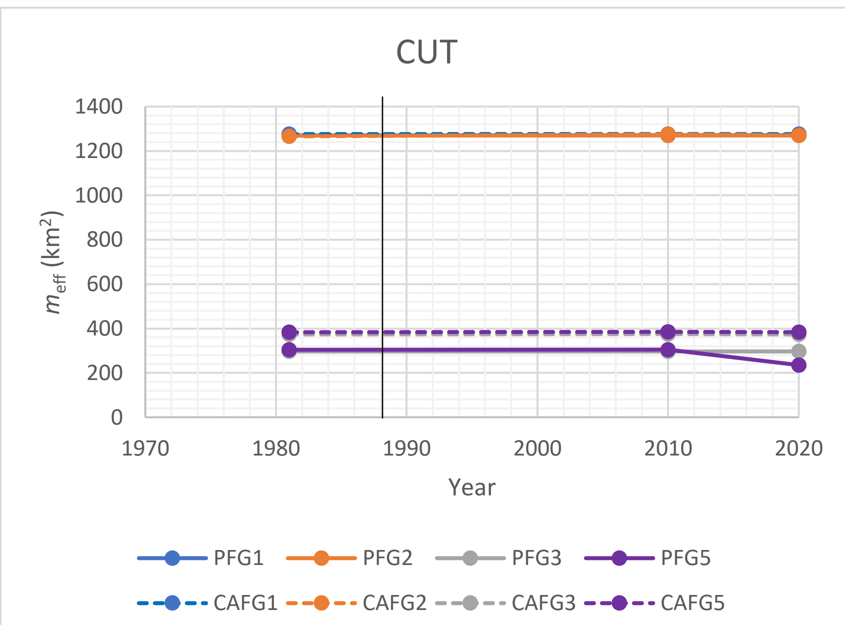


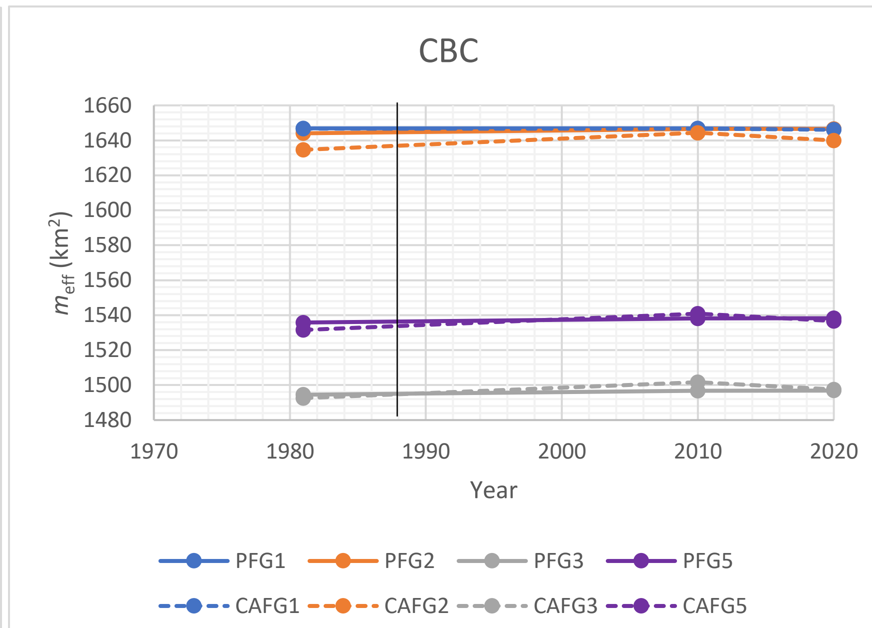
Gulf Islands, designated 2003



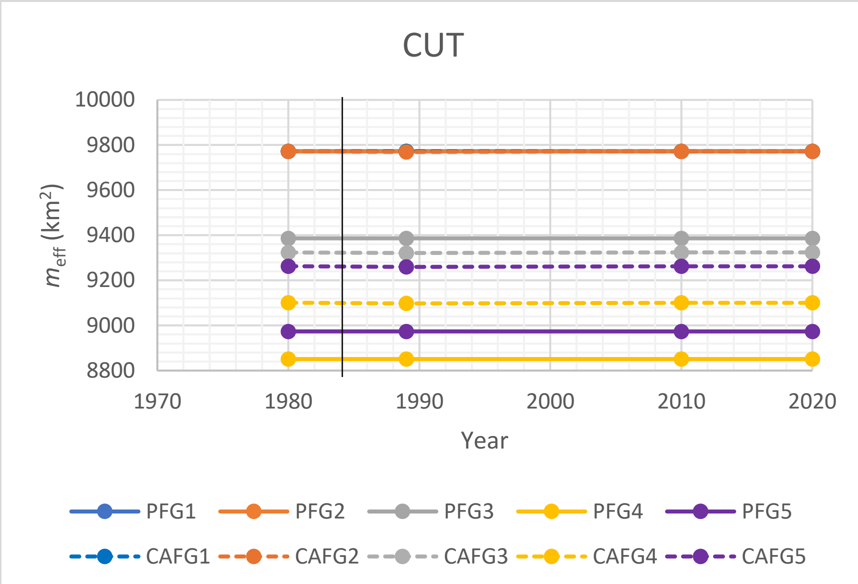


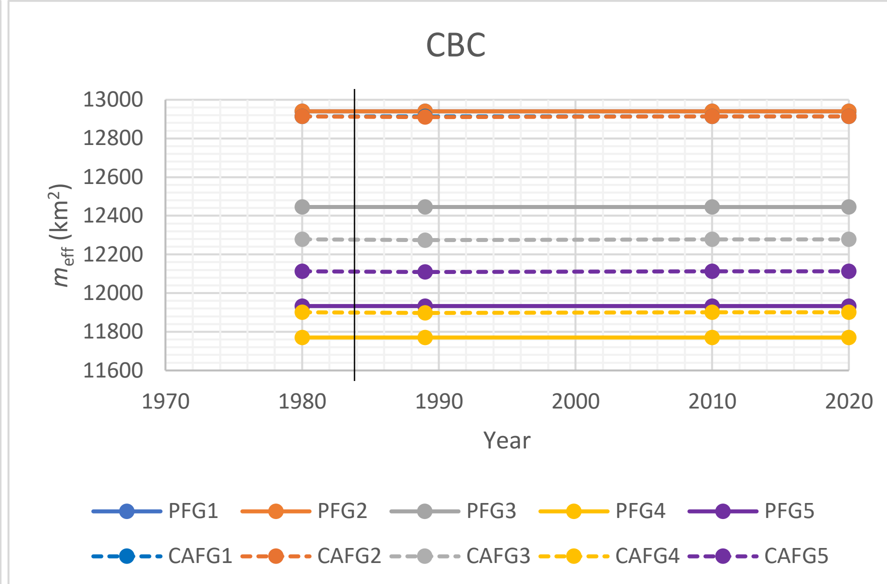
Gwaii Haanas, designated 1988



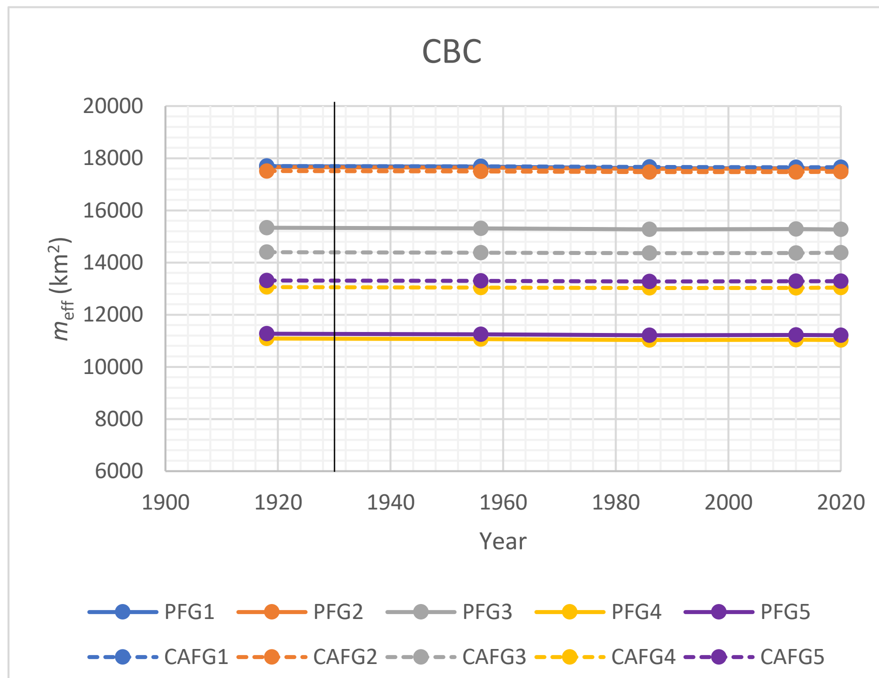


Ivvavik, designated 1984

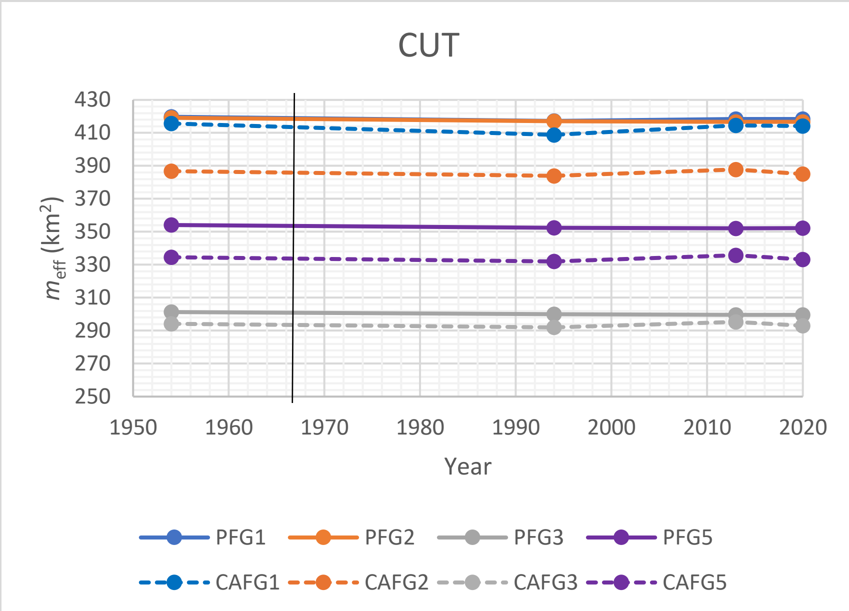


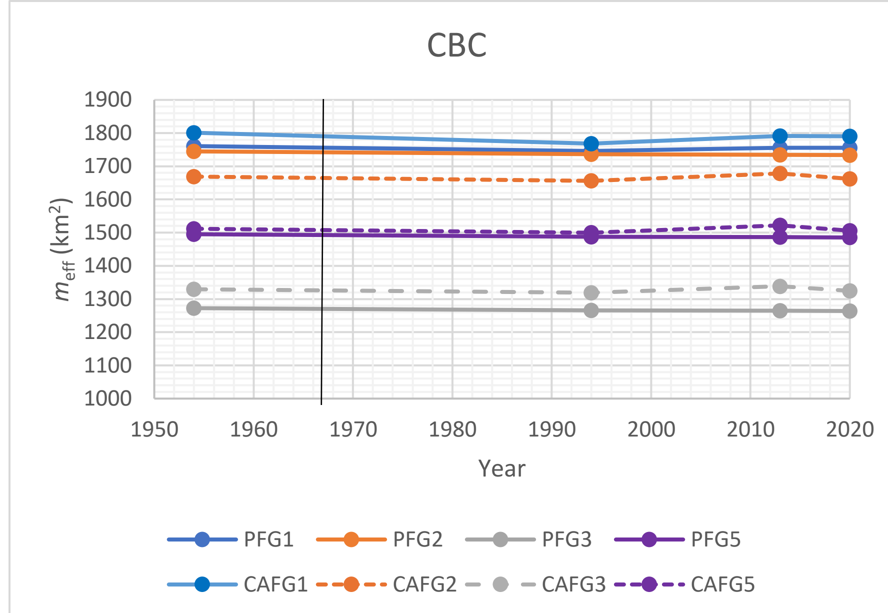


Jasper, designated 1907

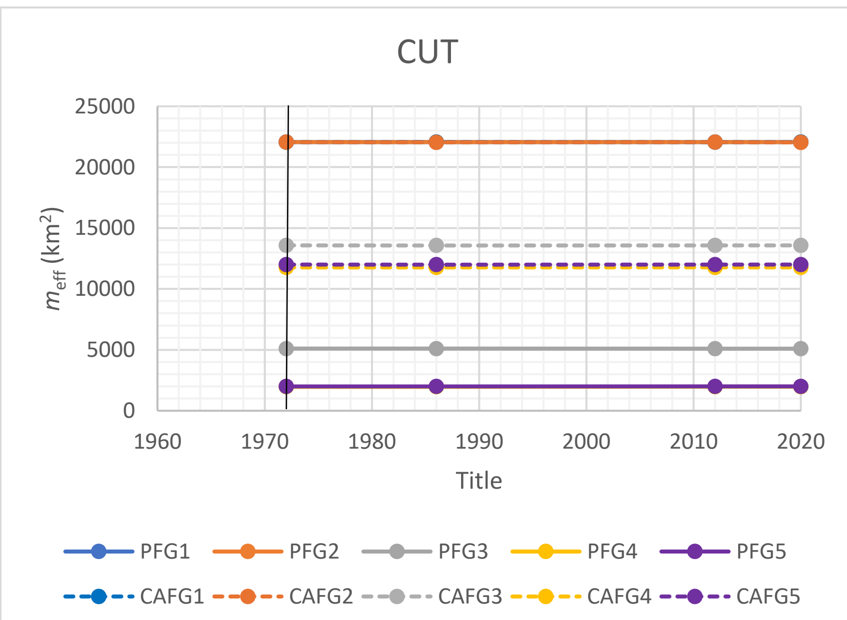


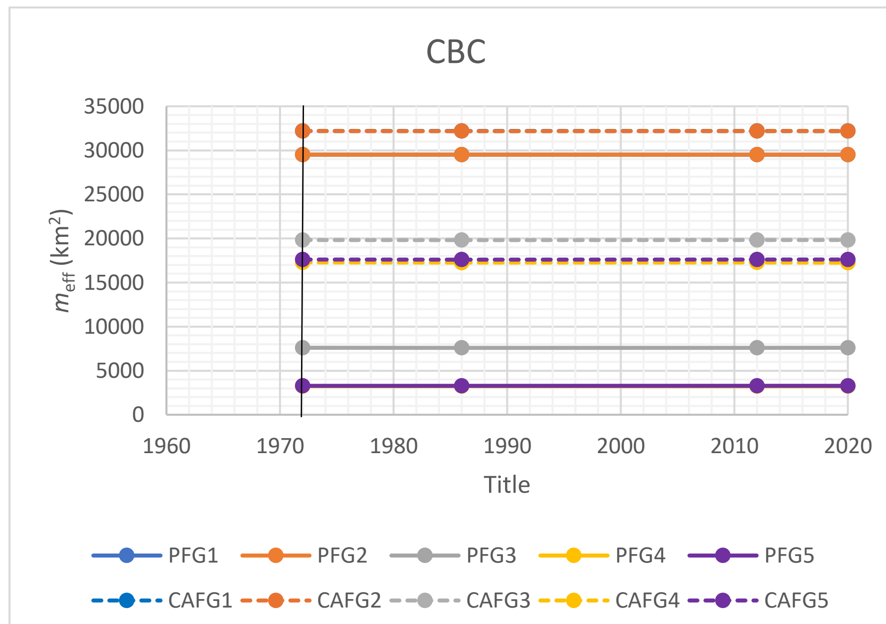
Kejimkujik, designated 1967



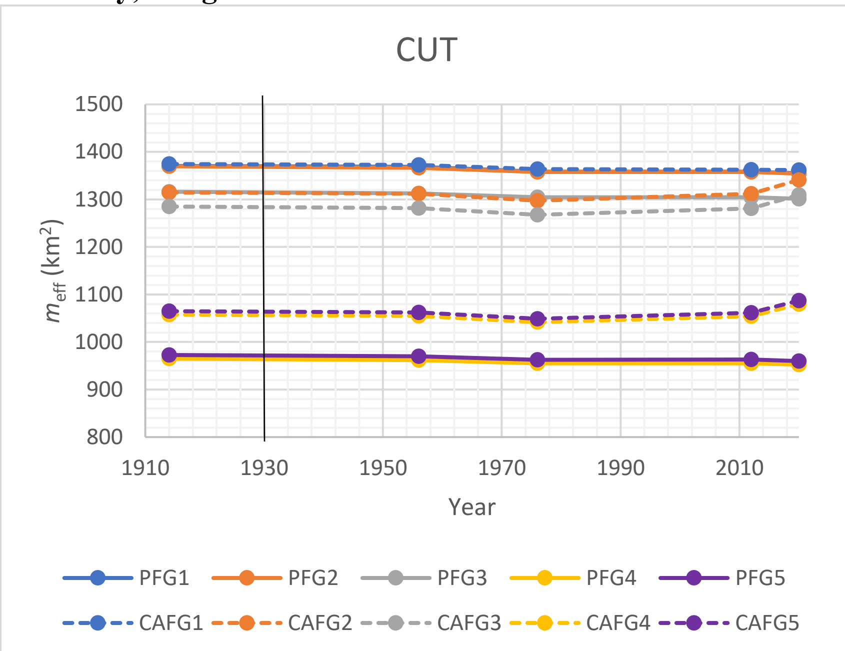


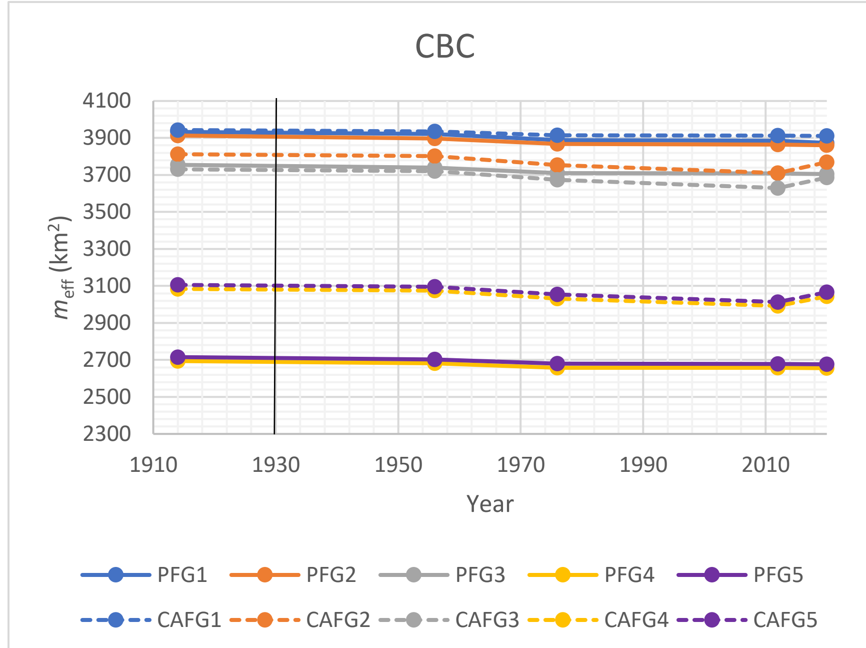
Kluane, designated 1972



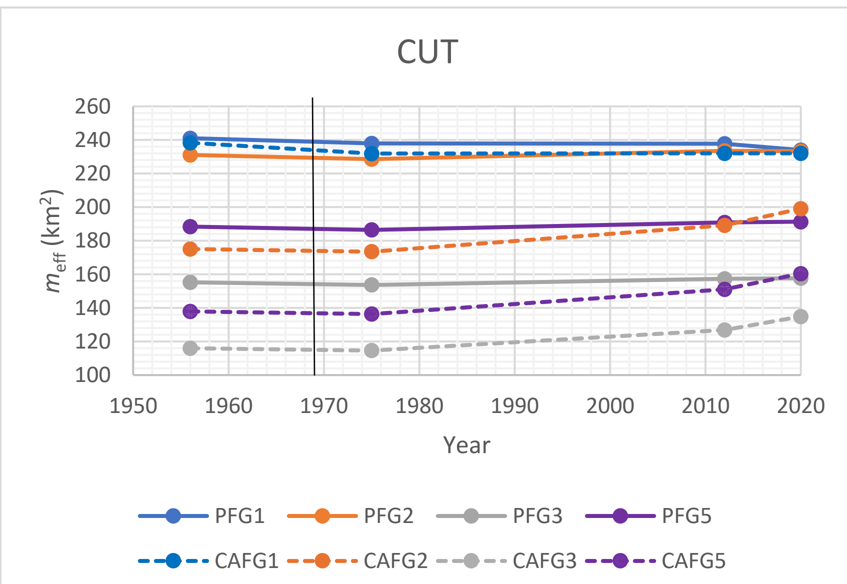


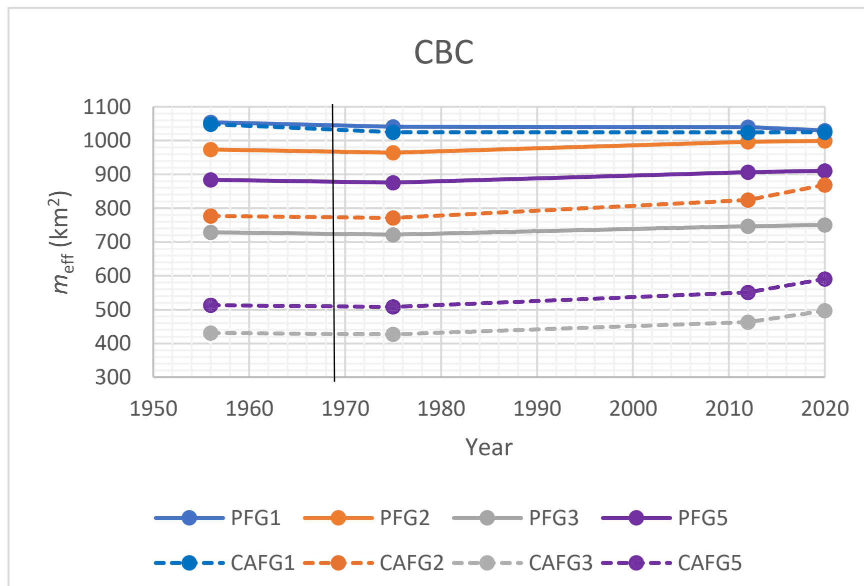
Kootenay, designated 1920



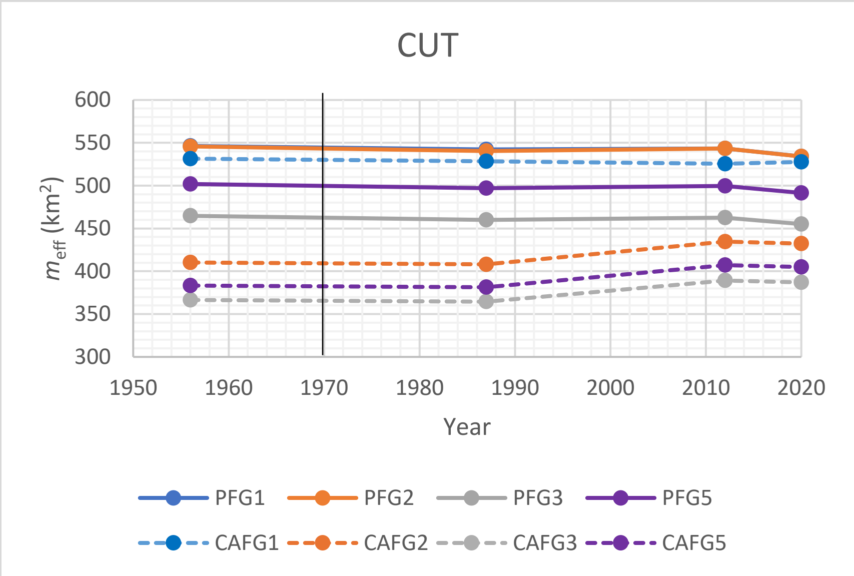


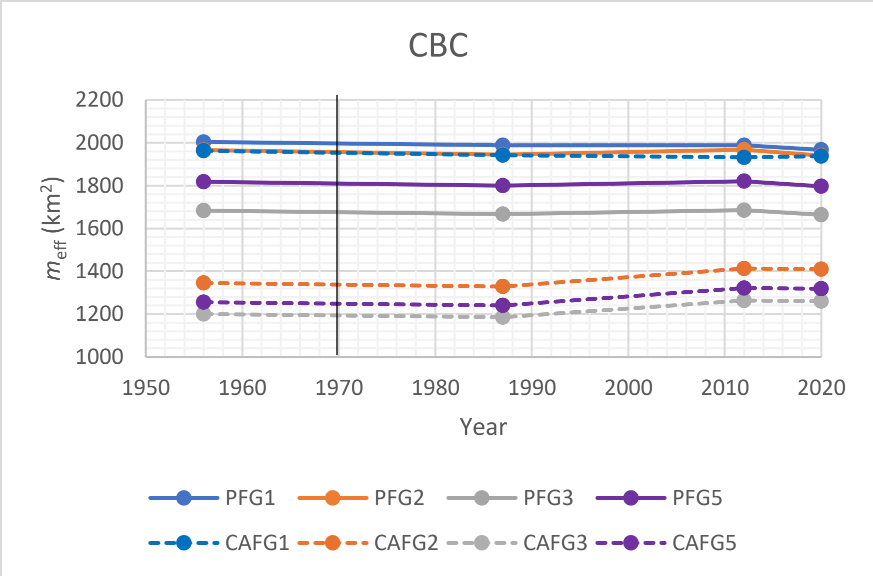
Kouchibouguac, designated 1969



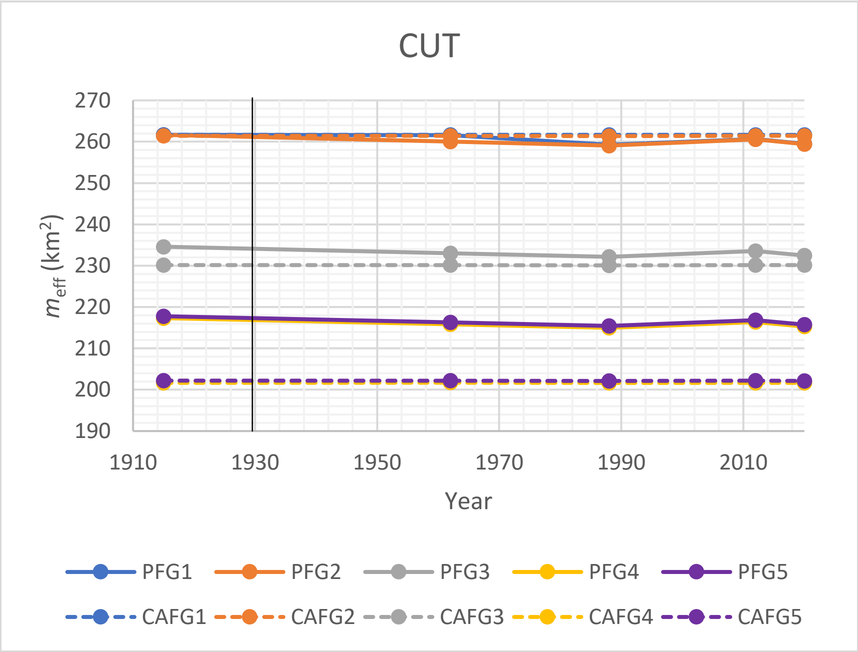


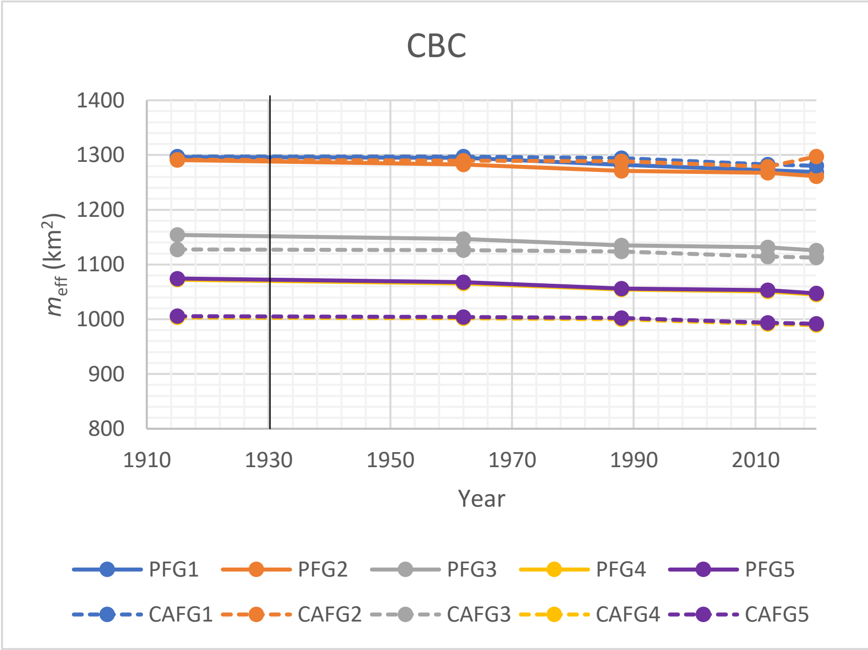
La Mauricie, designated 1970



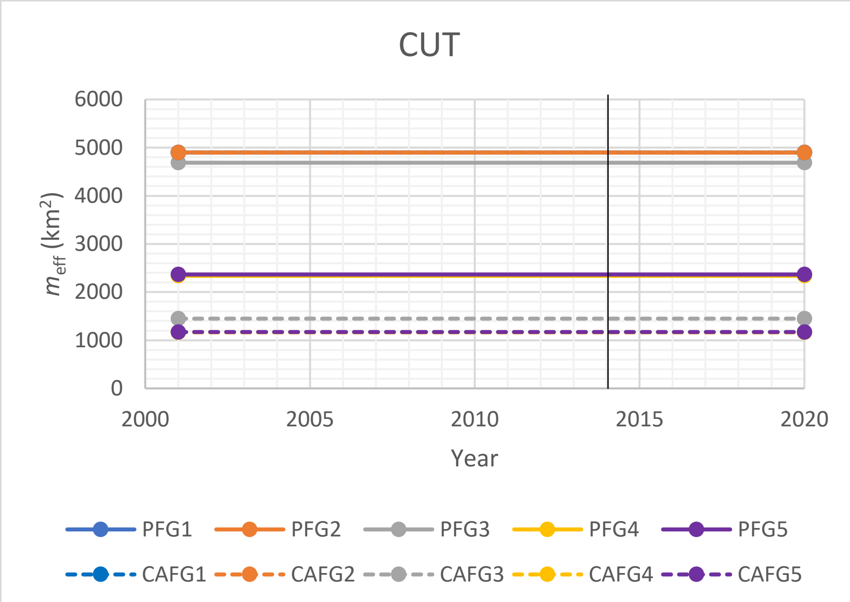


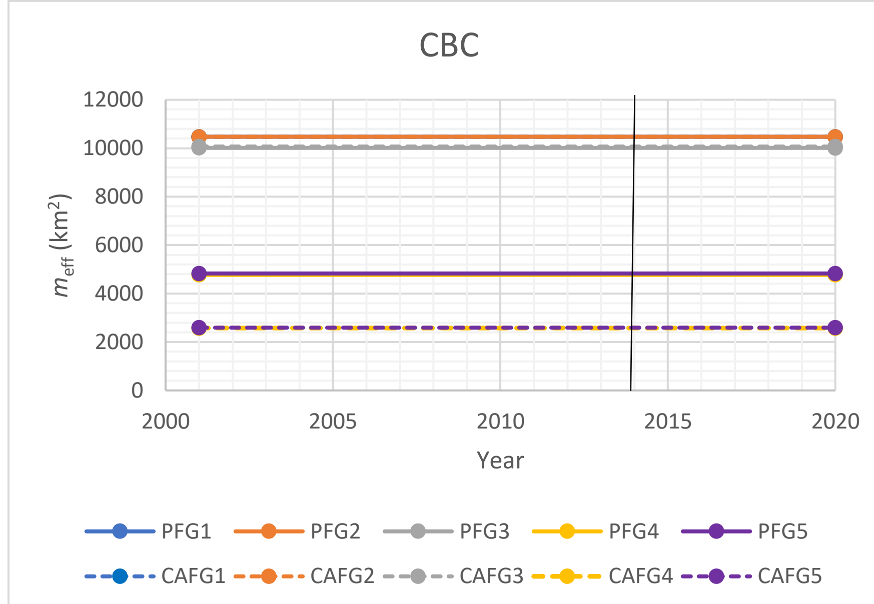
Mount Revelstoke, designated 1914



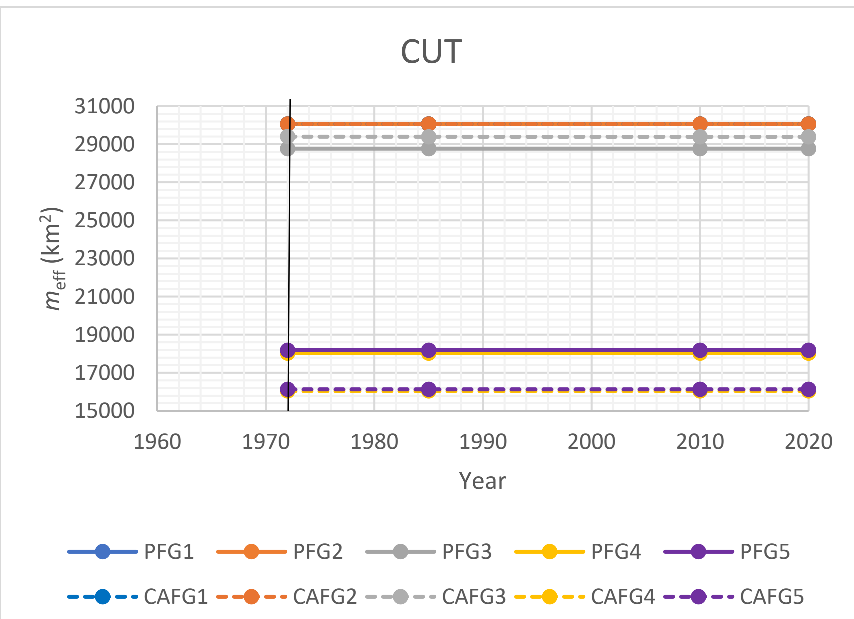


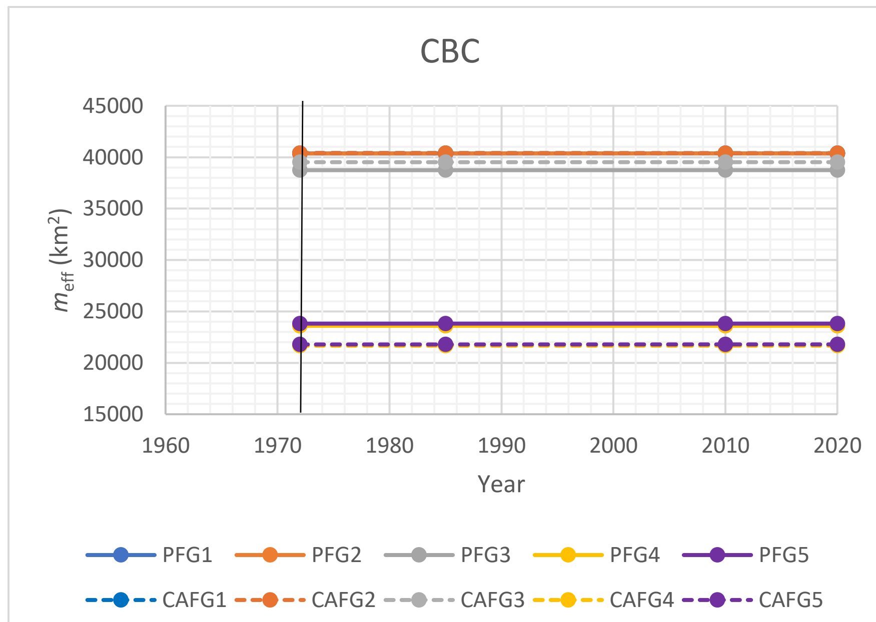
Nááts'įhch'oh, designated 2014



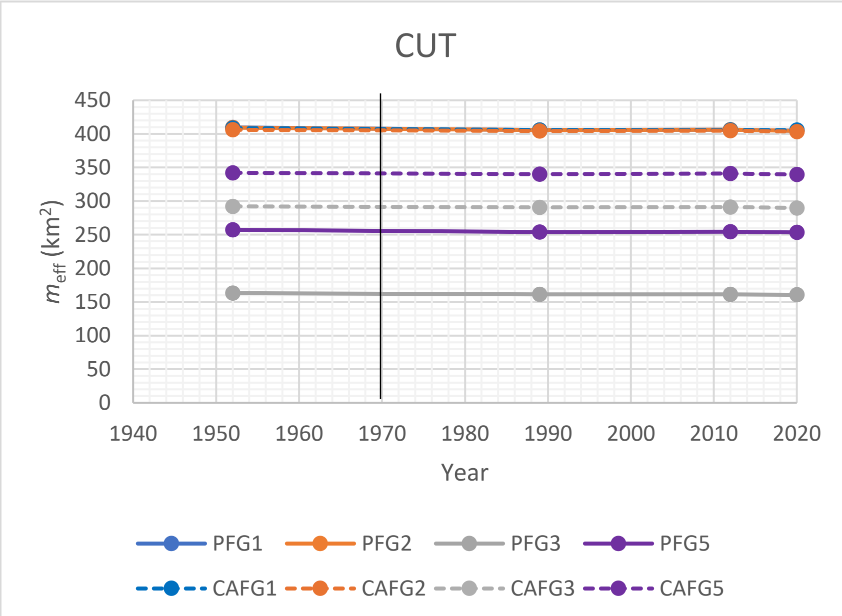


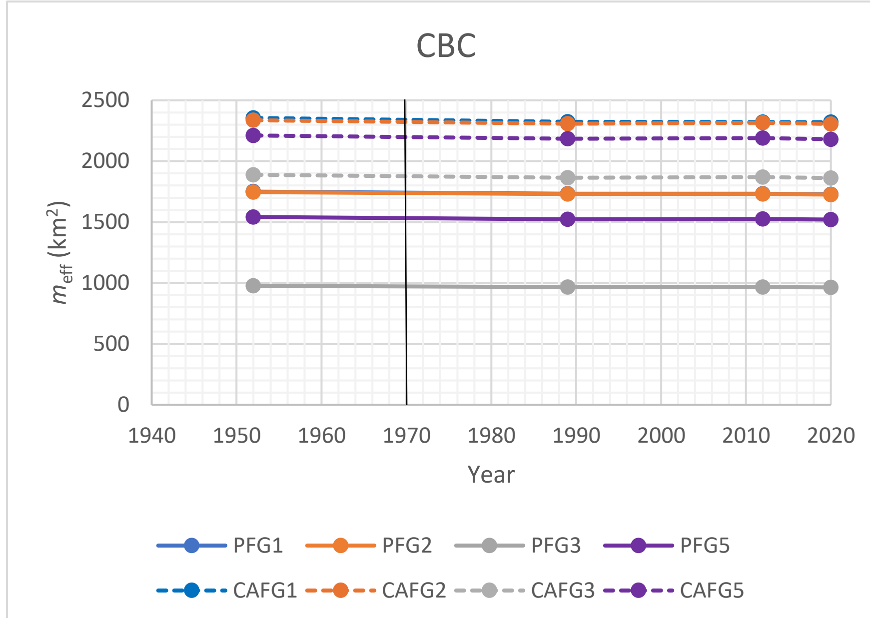
Nahanni, designated 1972



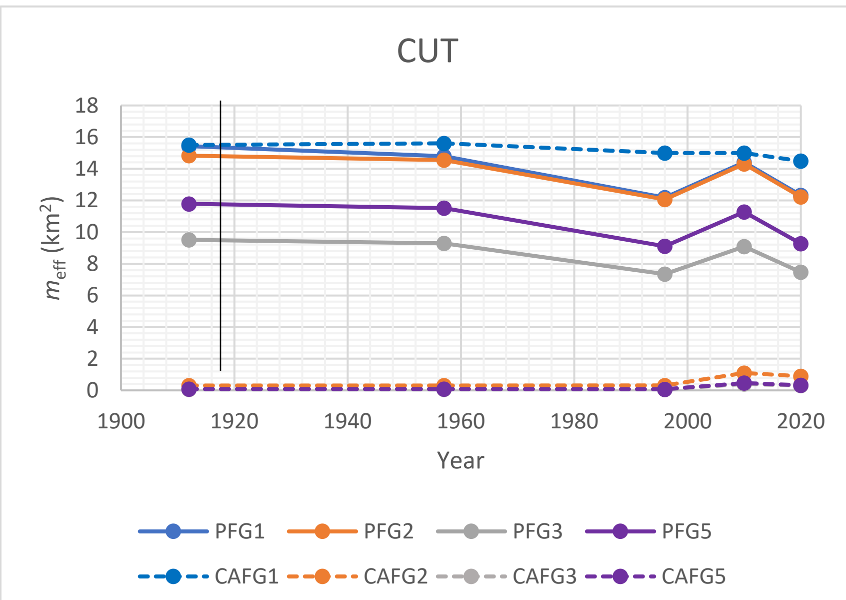


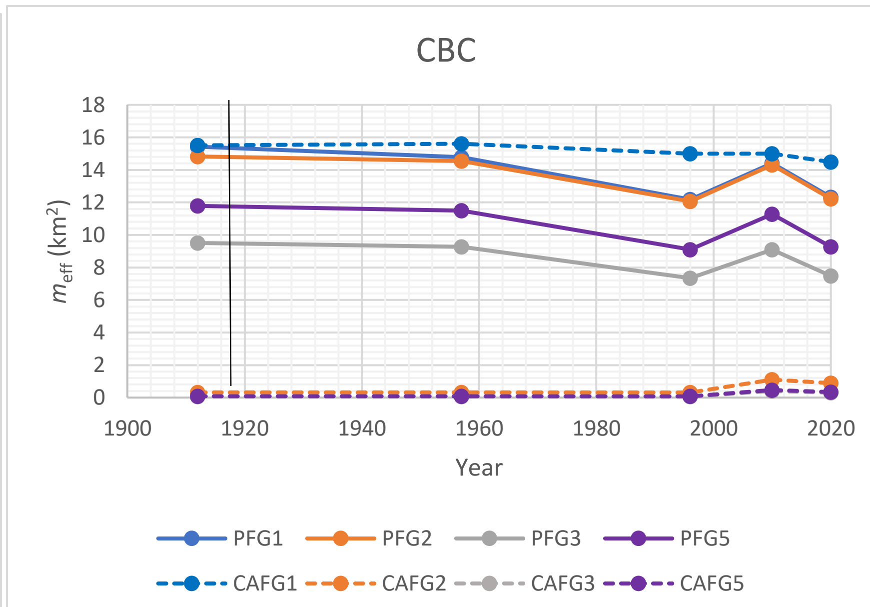
Pacific Rim, designated 1970



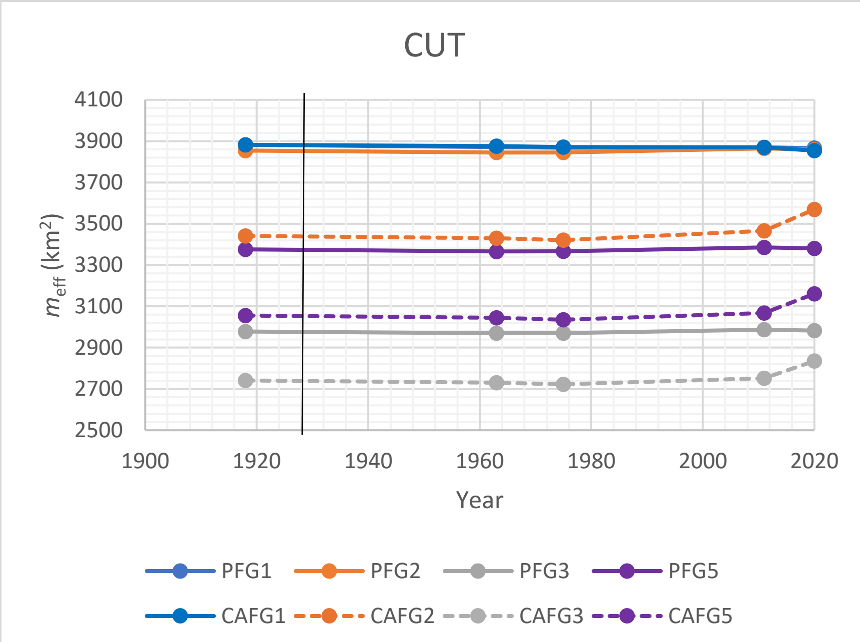


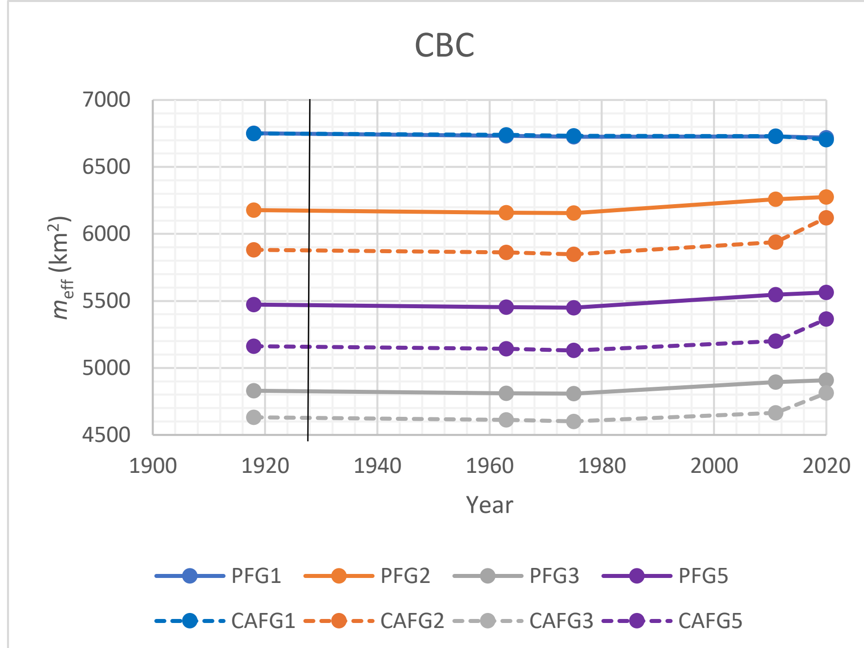
Point Pelee, designated 1918



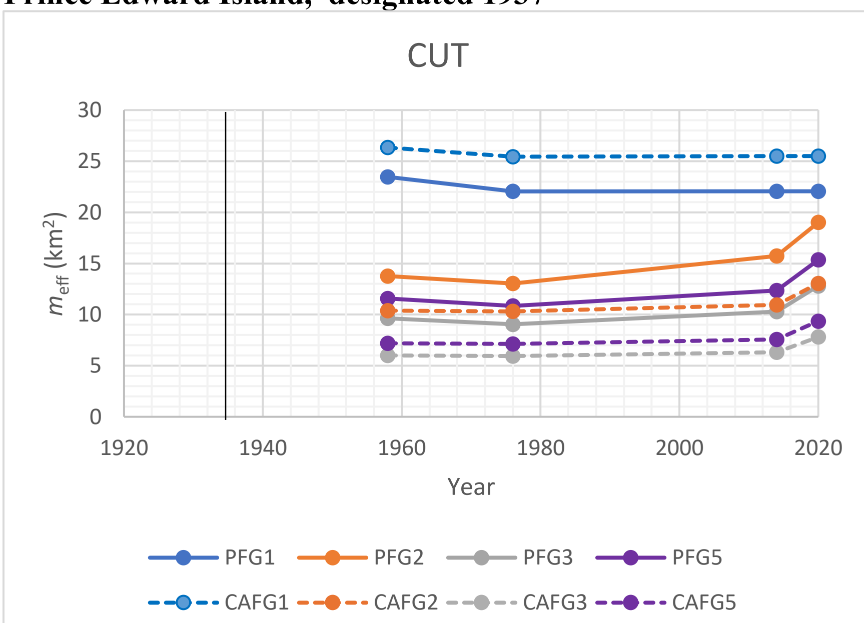


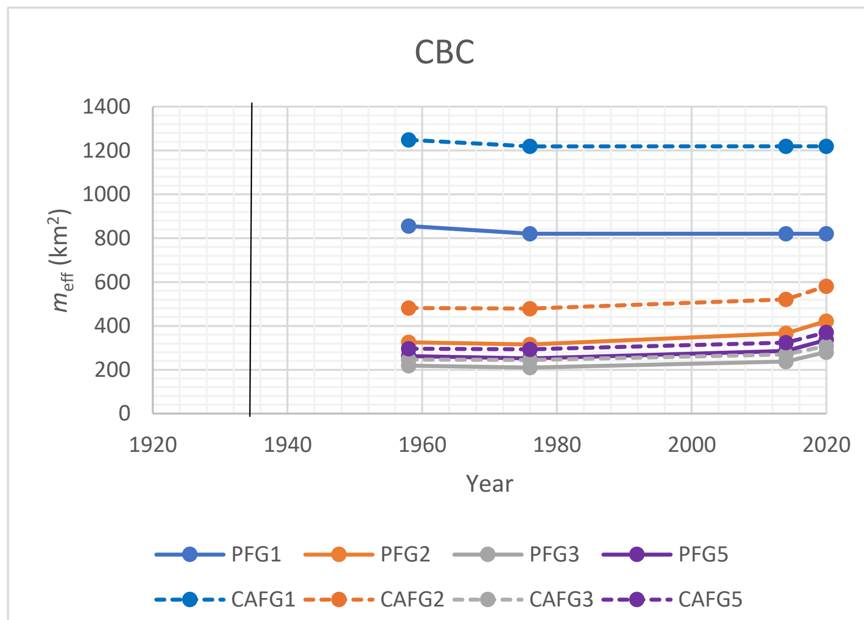
Prince Albert, designated 1927



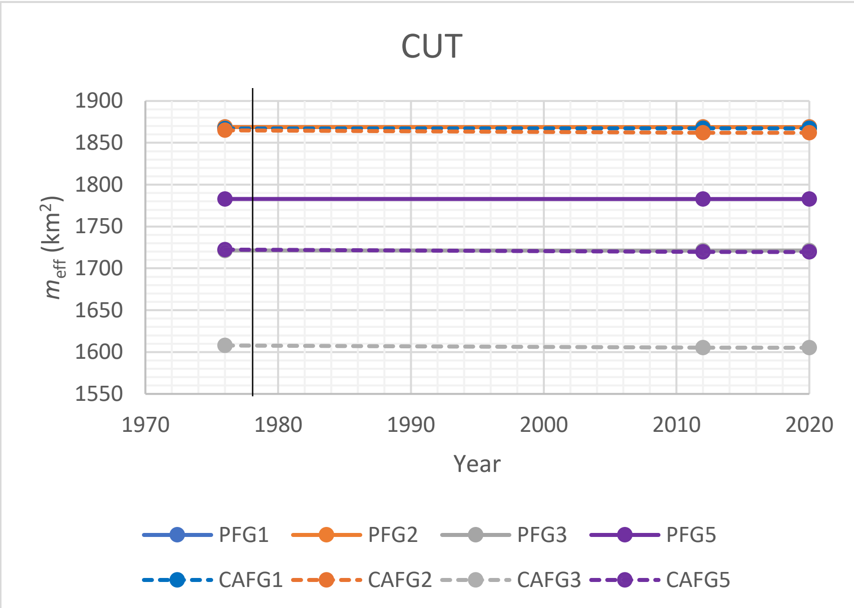


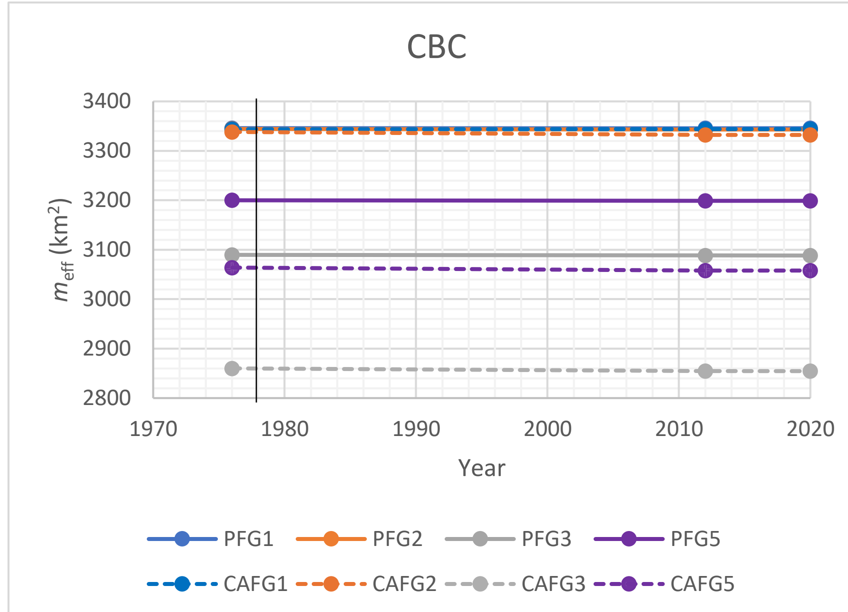
Prince Edward Island, designated 1937



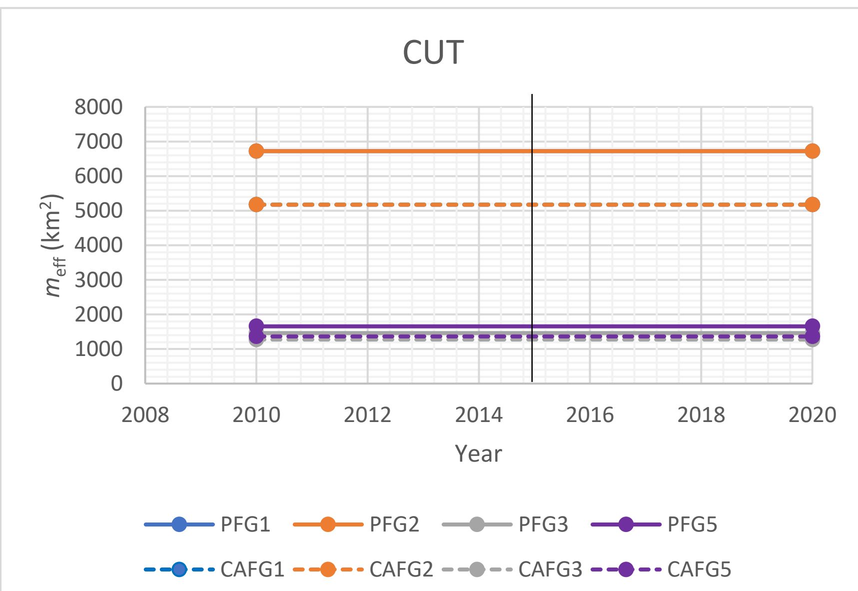


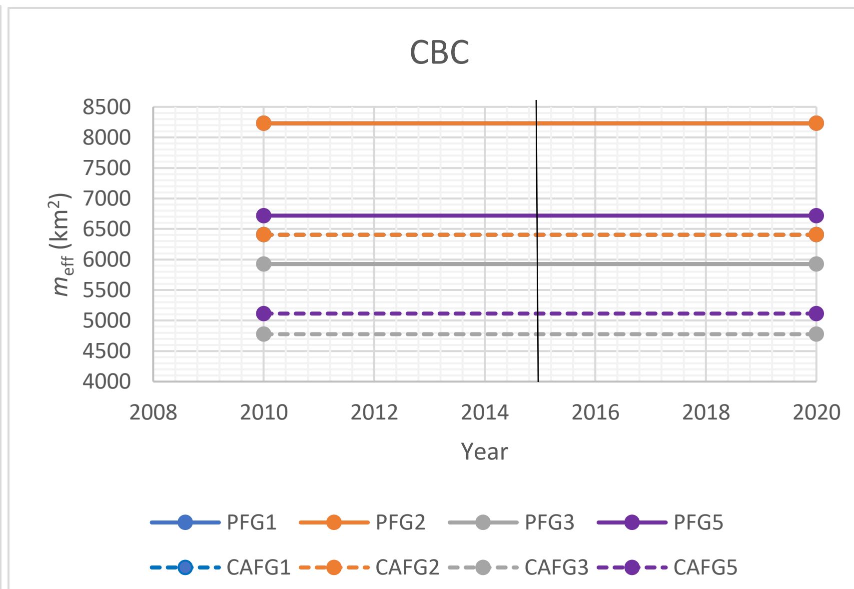
Pukaskwa, designated 1978



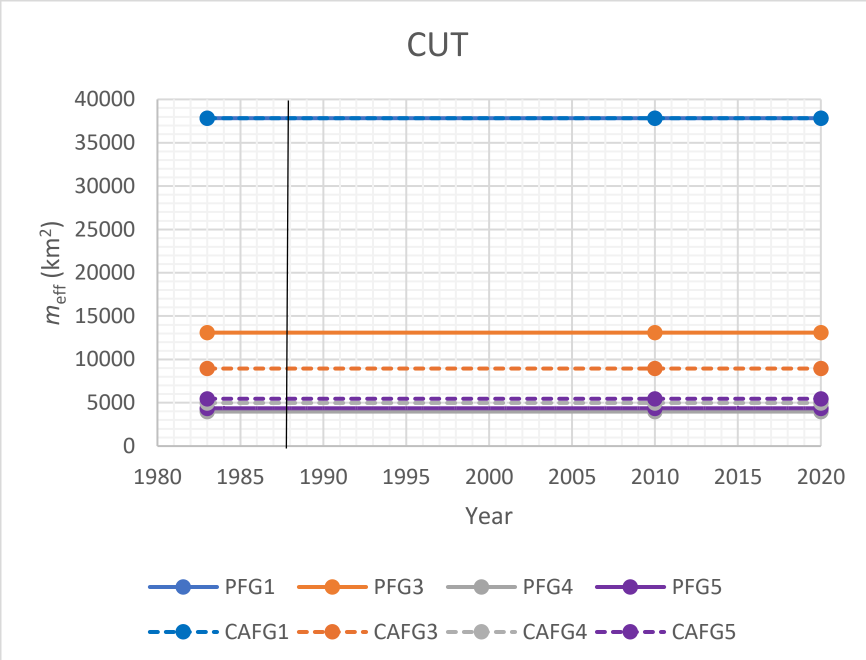


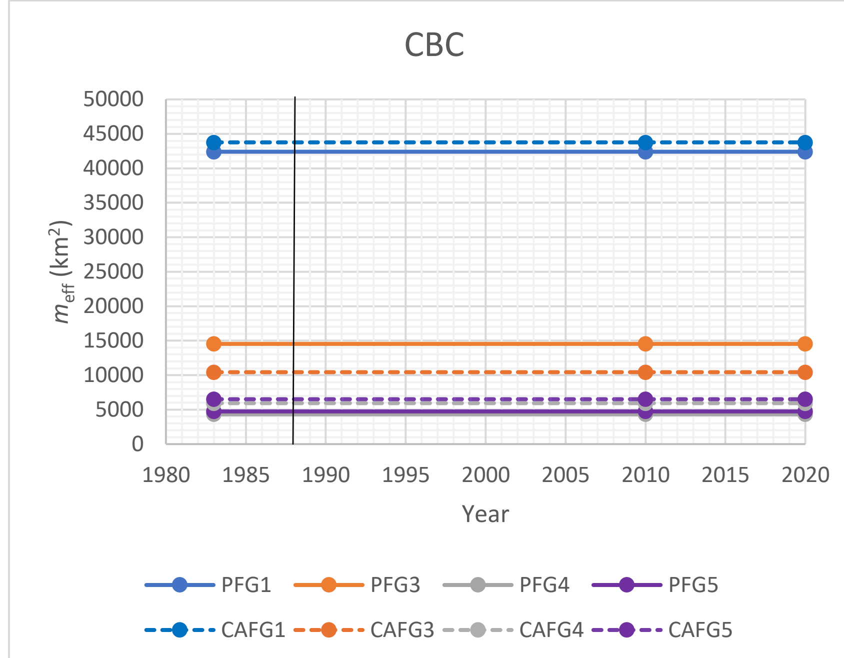
Qausuittuq, designated 2015



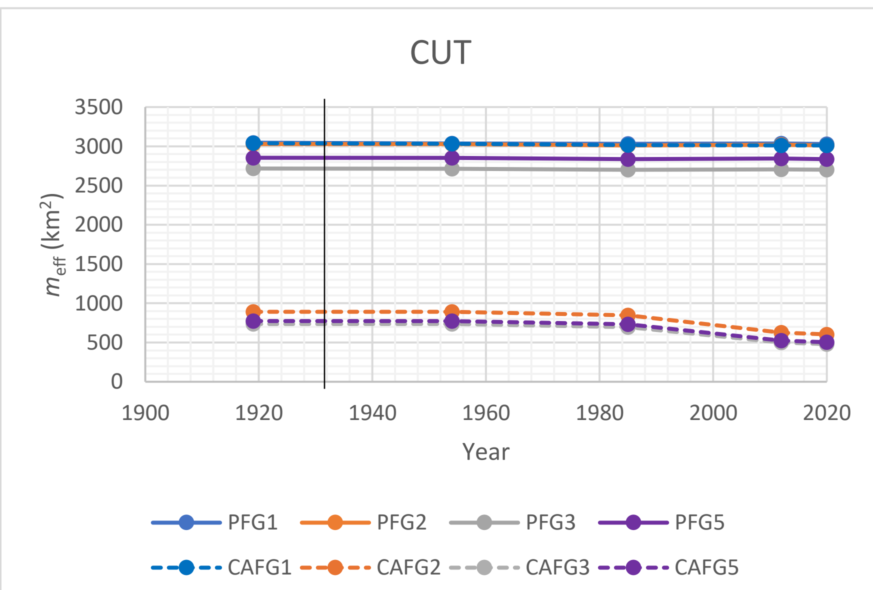


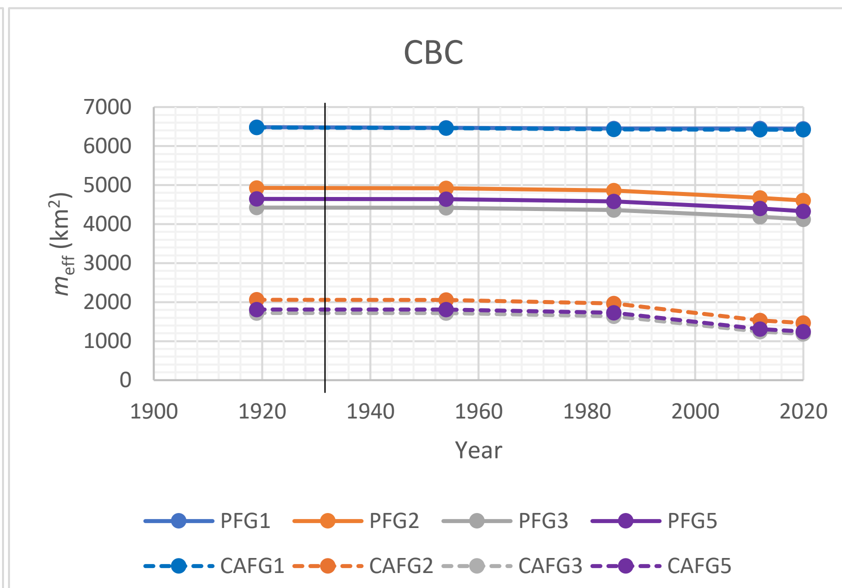
Quttinirpaaq, designated 1988



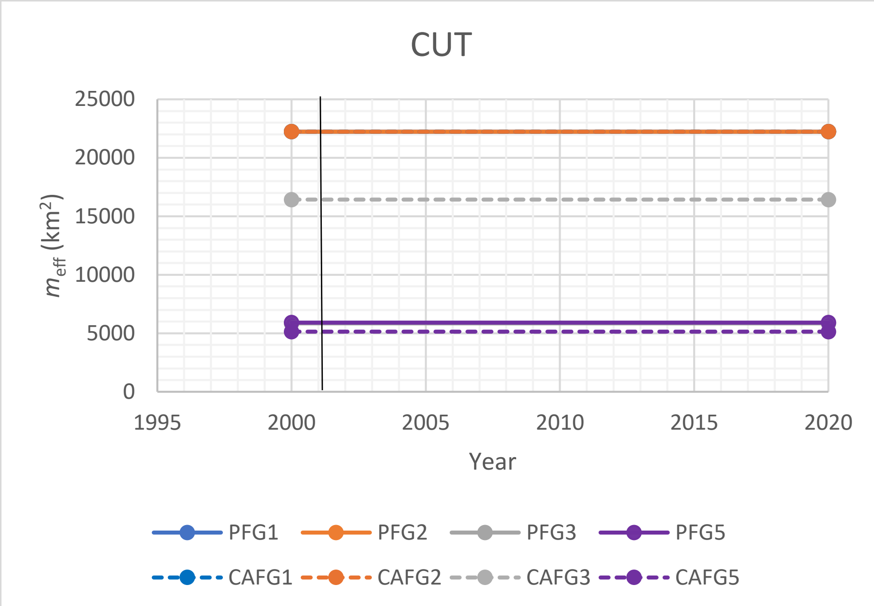


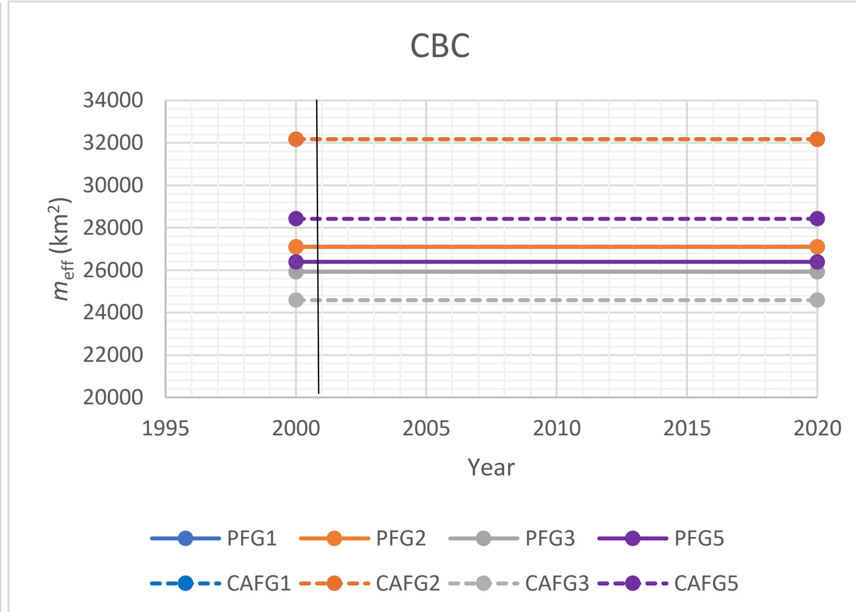
Riding Mountain, designated 1933



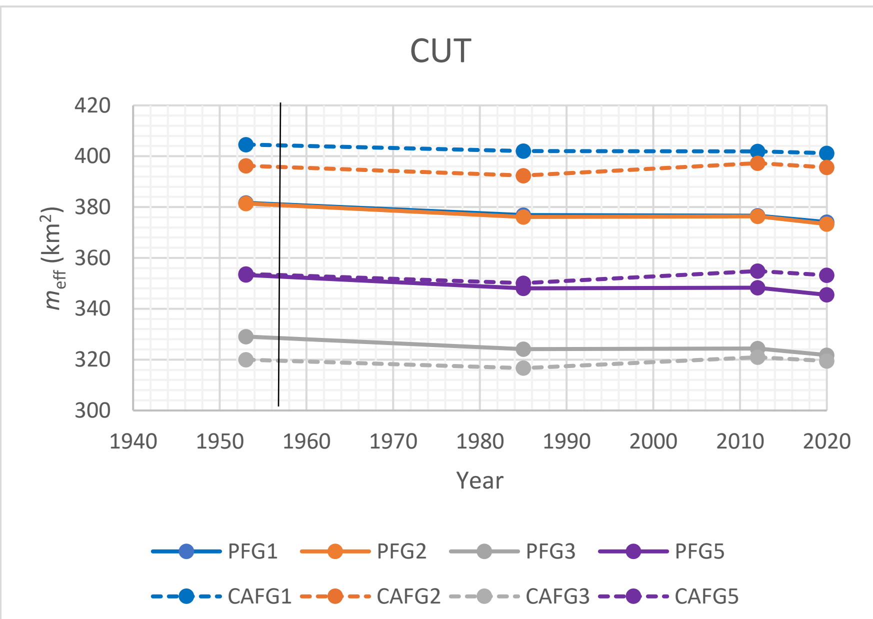


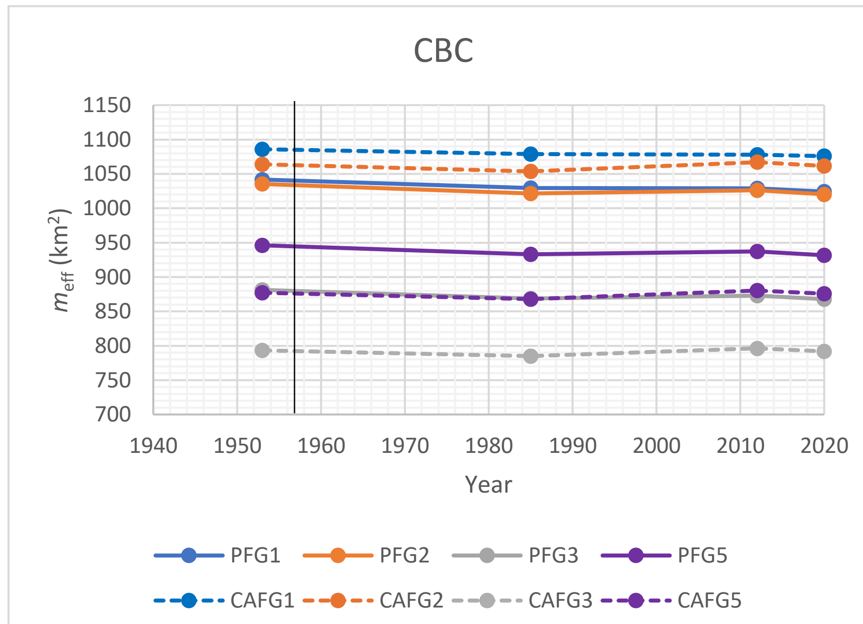
Sirmilik, designated 2001



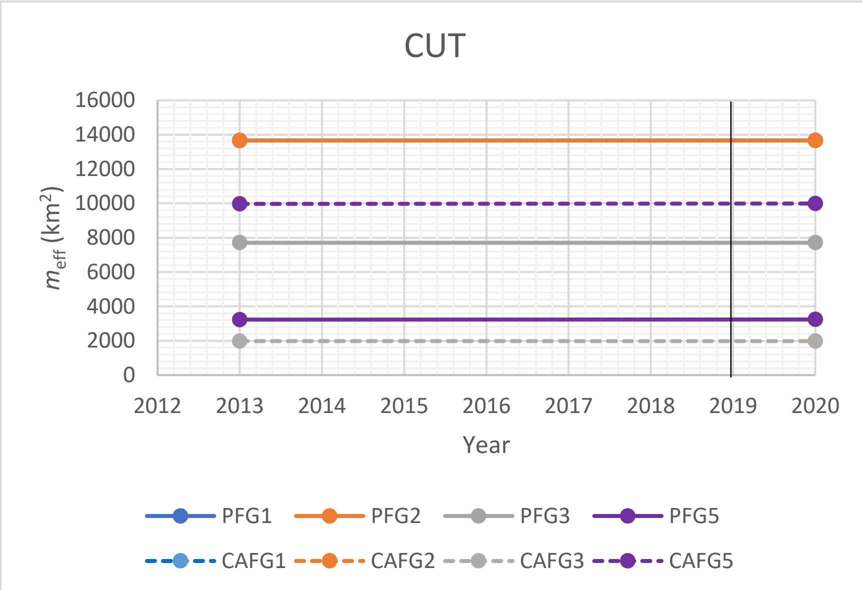


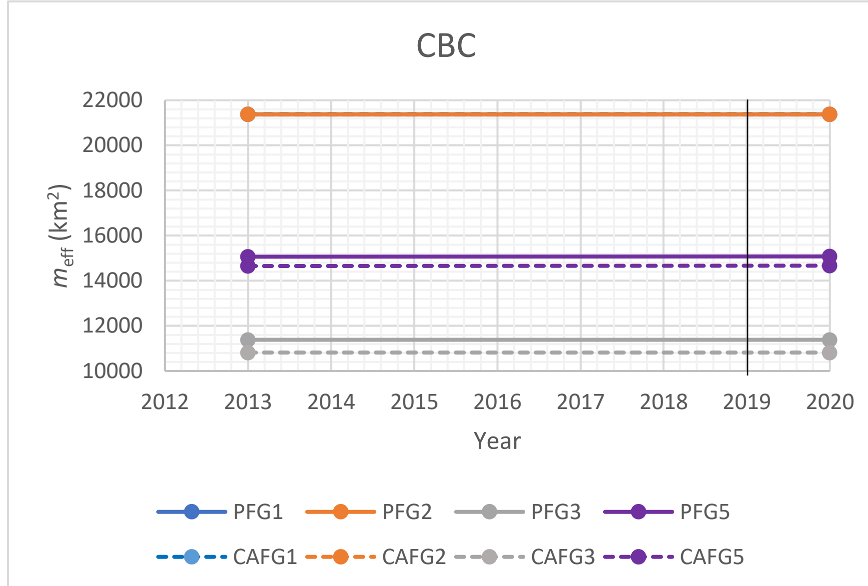
Terra Nova, designated 1957



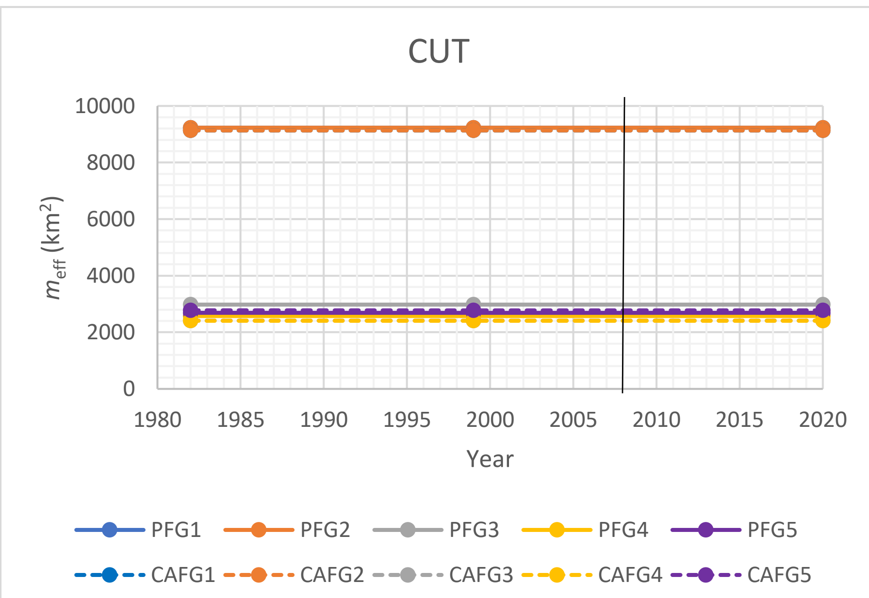


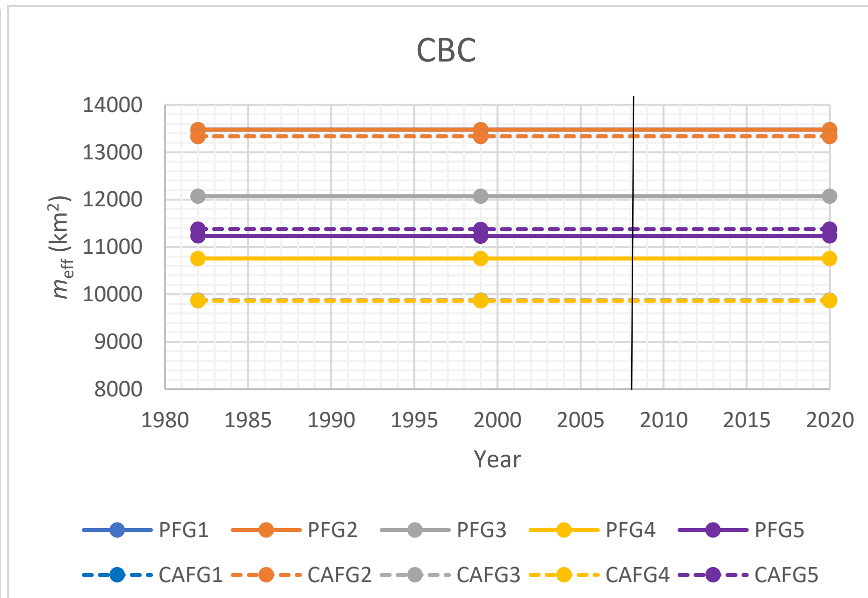
Thaidene Nëné, designated 2019



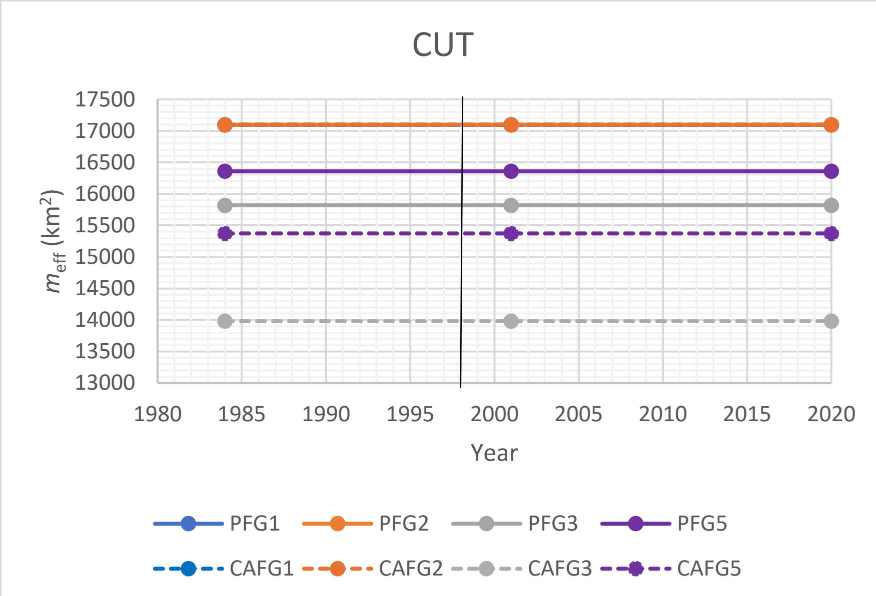


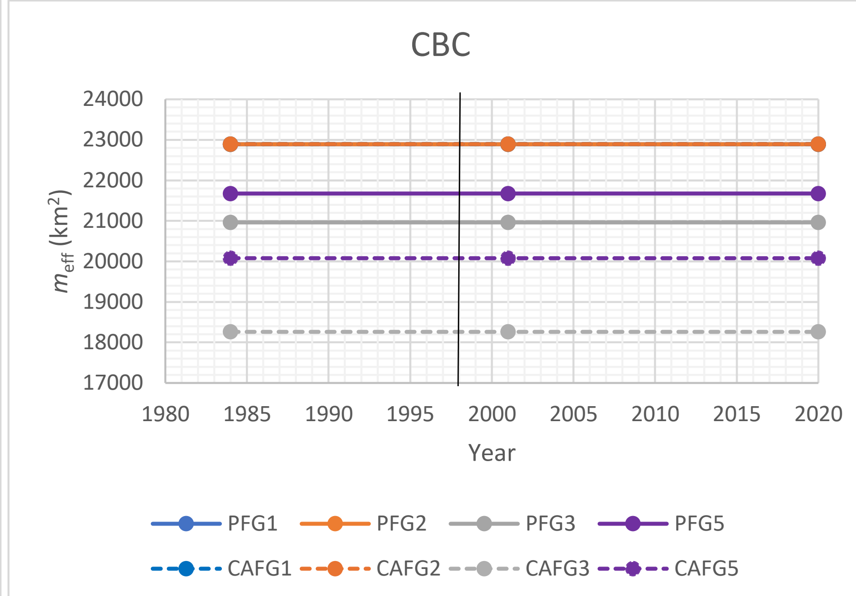
Tongait KakKasuangita SilakKijapvinga — Torngat Mountains, designated 2008



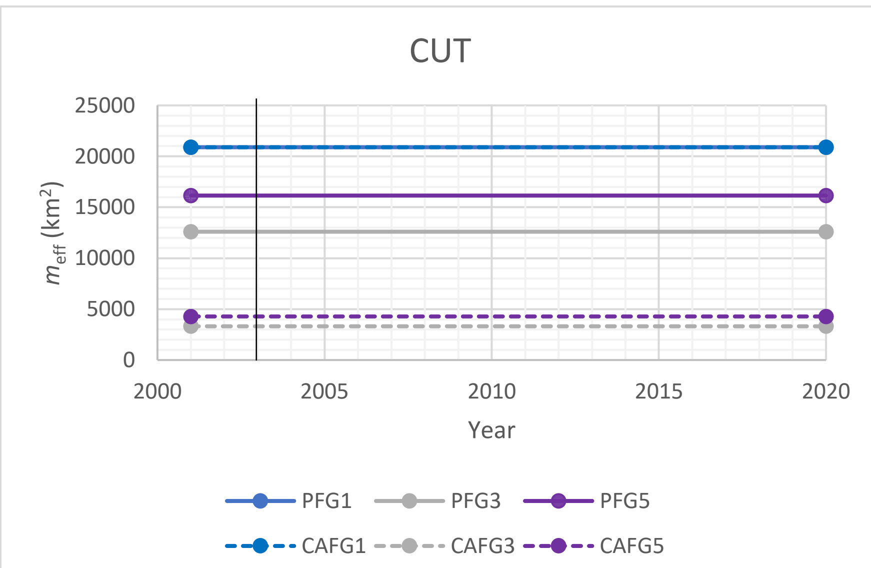


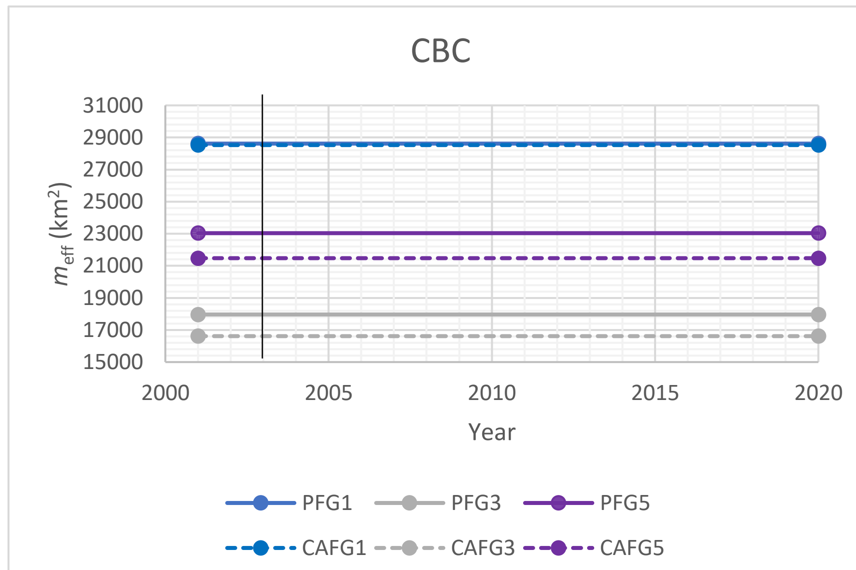
Tuktut Nogait, designated 1998



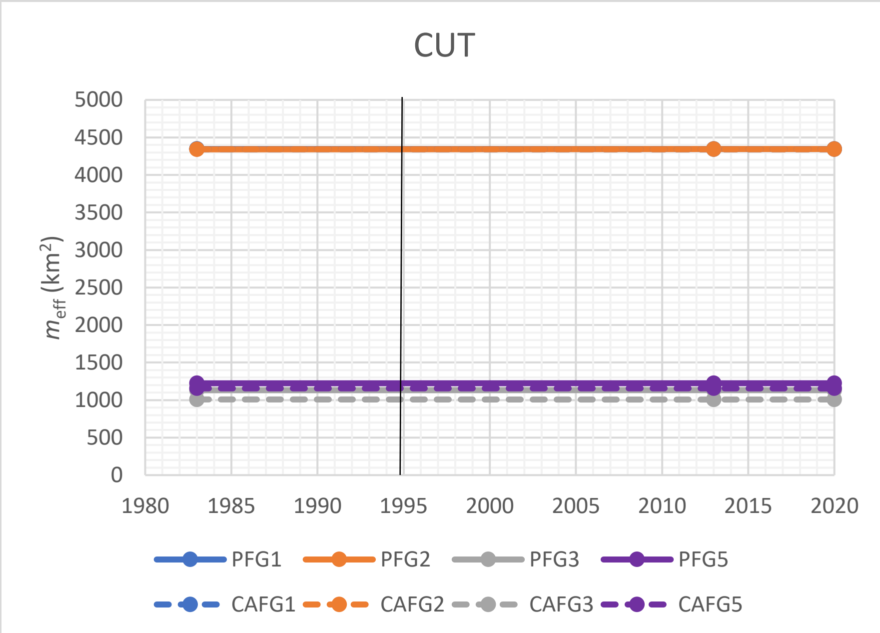


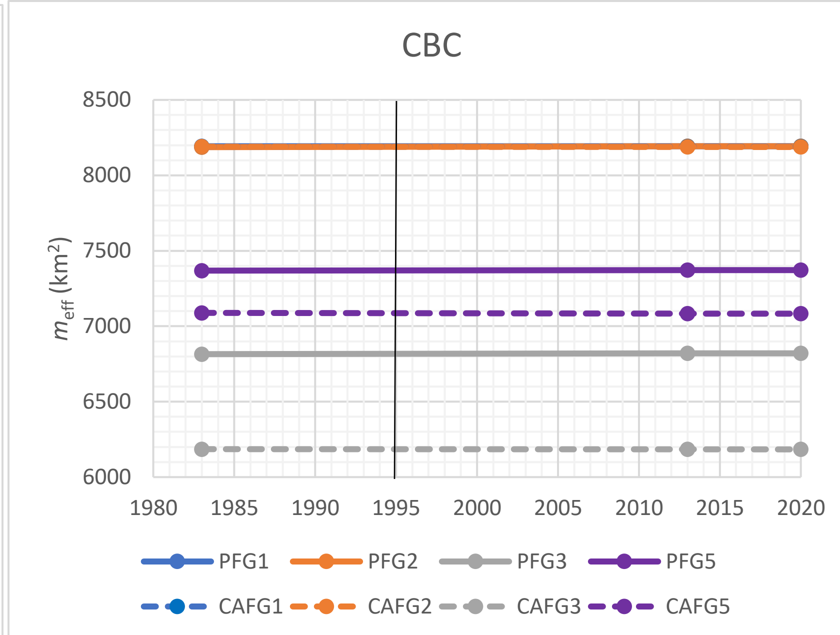
Ukkusiksalik, designated 2003



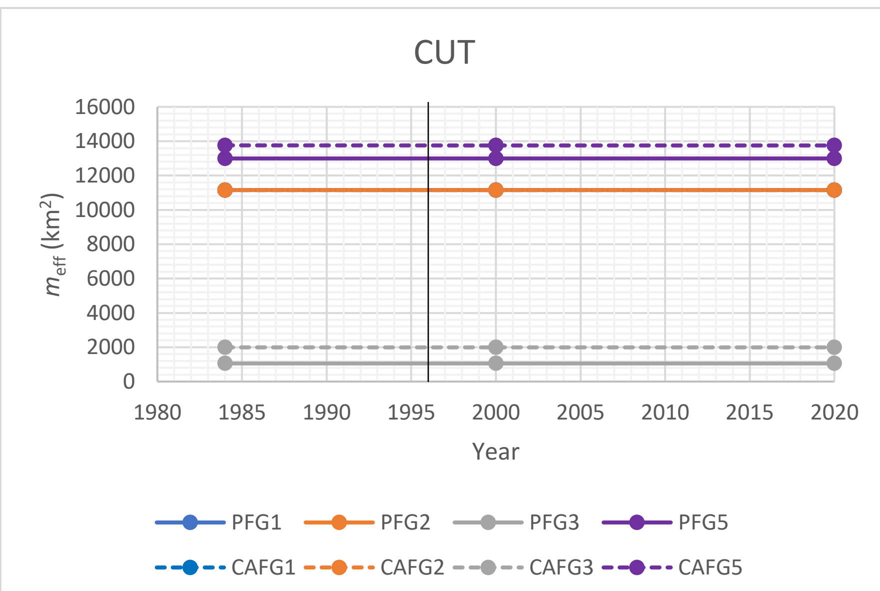


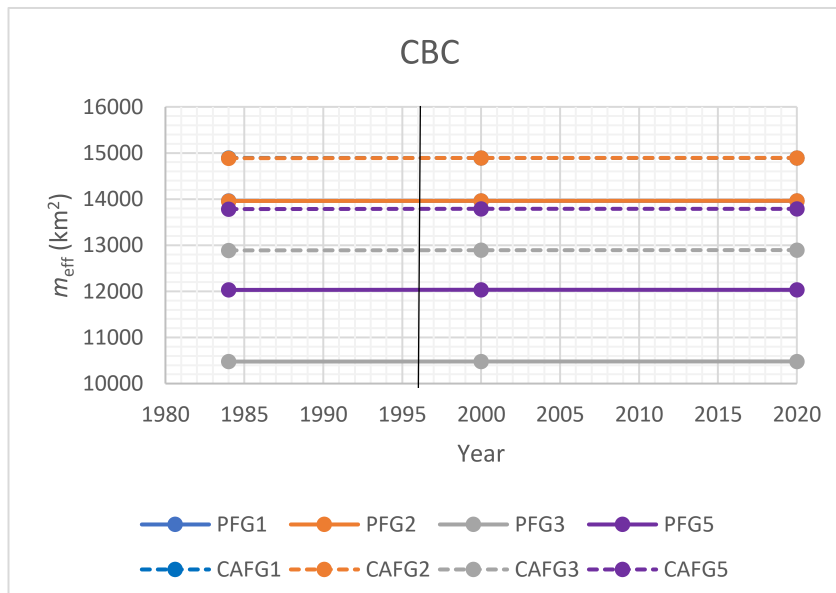
Vuntut, designated 1995



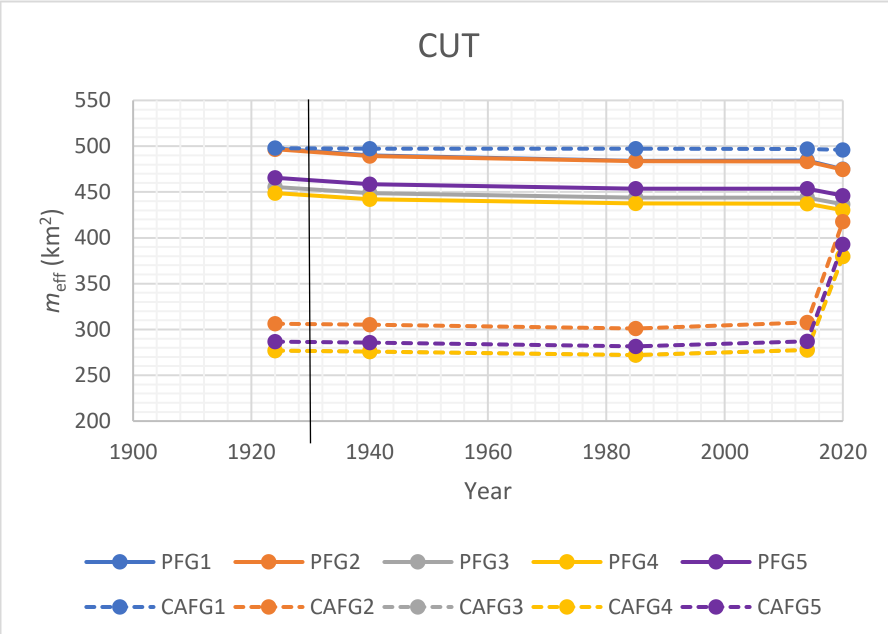


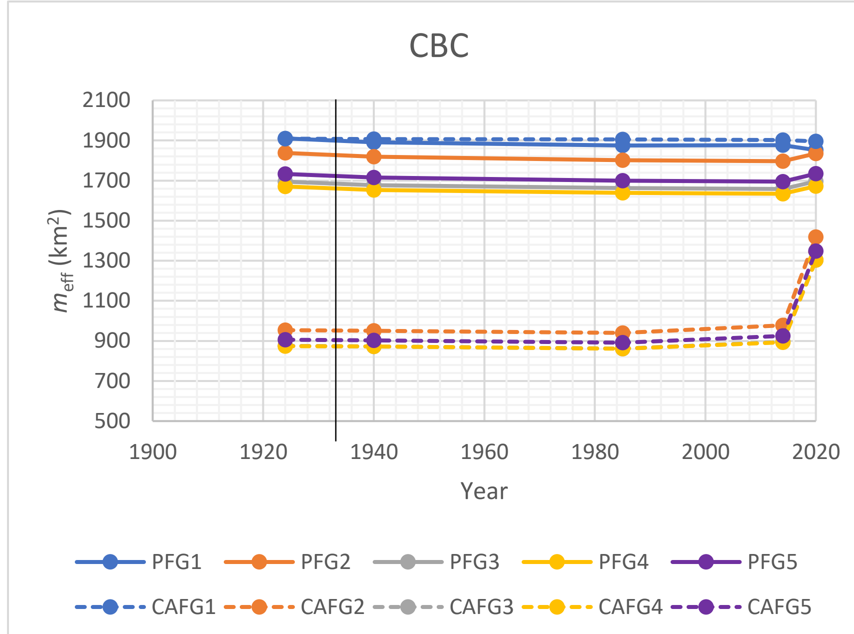
Wapusk, designated 1996



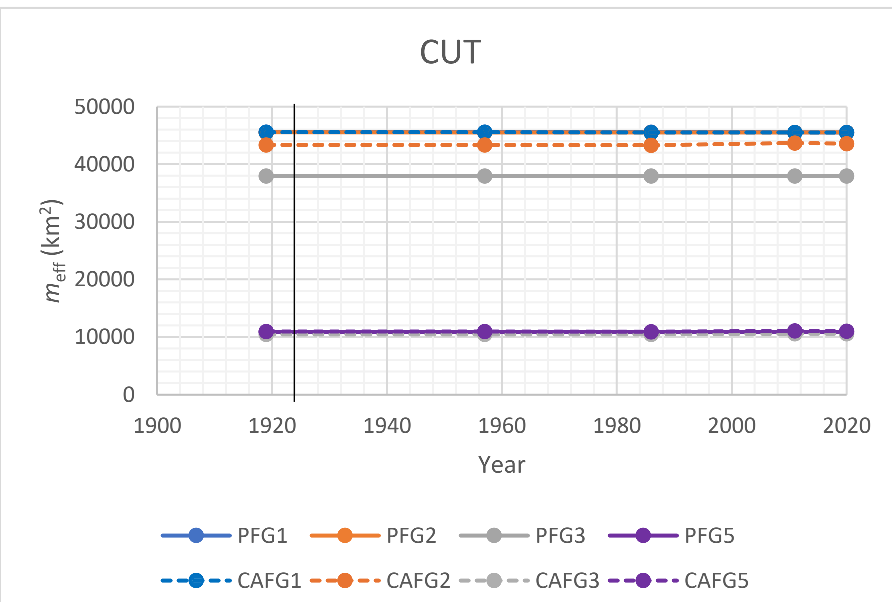


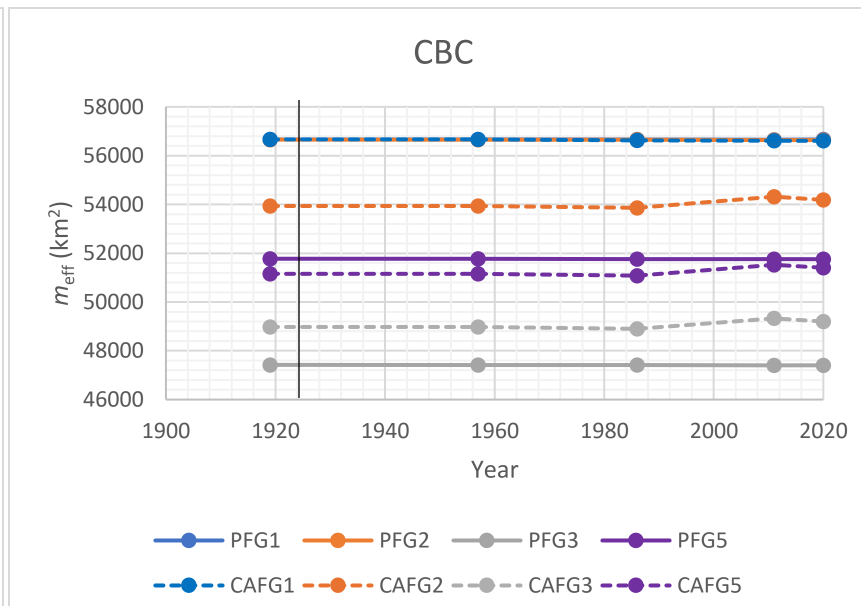
Waterton Lakes, designated 1885



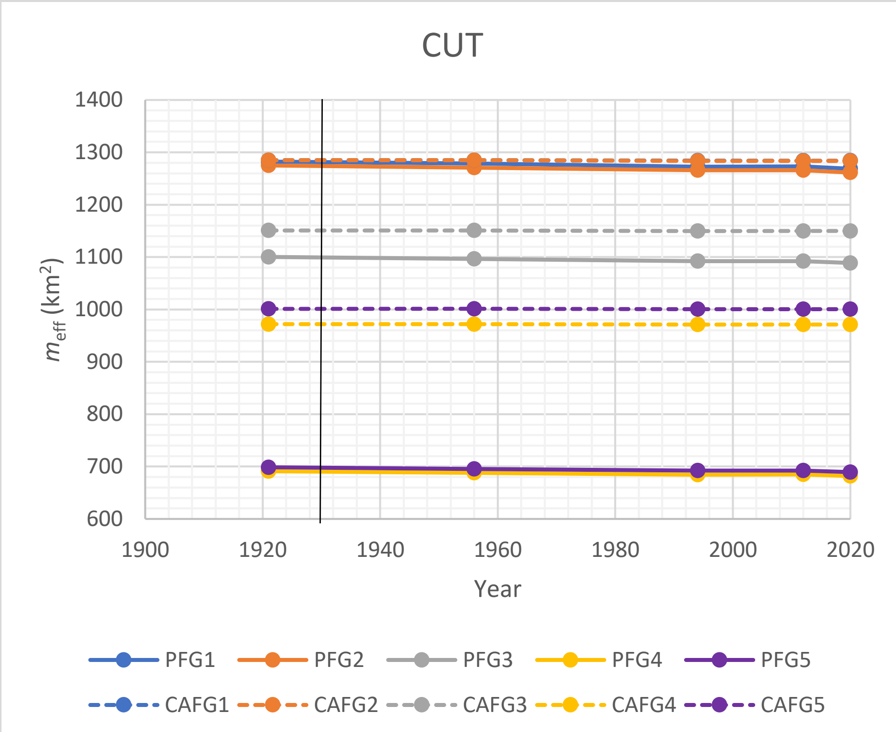


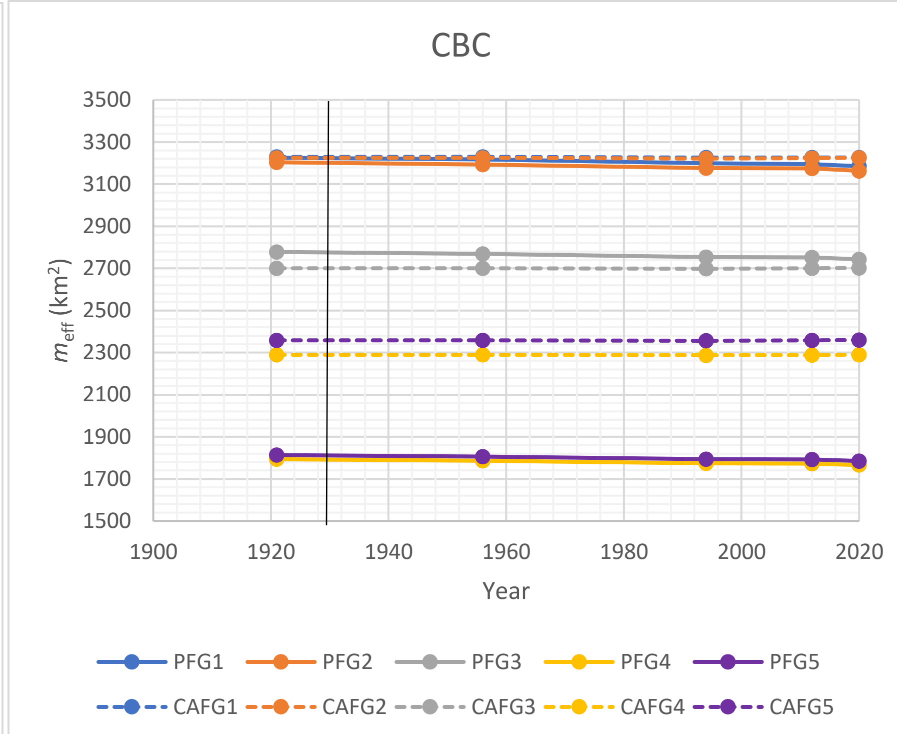
Wood Buffalo, designated 1922





Yoho, designated 1886





6.3 Data used in study

6.3.1 Vector datasets

Name	
National Park boundaries	Parks Canada Agency, G. of C. (2020). *Places administered by Parks Canada—Open Government Portal* (Version 1) [.shp]. https://open.canada.ca/data/en/dataset/e1f0c975-f40c-4313-9be2-beb951e35f4e
CanVec – 50k topographic	Natural Resources Canada. (2020). *Topographic Data of Canada—CanVec Series—Open Government Portal* [.shp]. https://ouvert.canada.ca/data/dataset/8ba2aa2a-7bb9-4448-b4d7-f164409fe056
CanVec – 250k hydrographic features	Natural Resources Canada. (2020). *Topographic Data of Canada—CanVec Series—Open Government Portal* [.shp]. https://ouvert.canada.ca/data/dataset/8ba2aa2a-7bb9-4448-b4d7-f164409fe056
Agriculture – Annual Crop Inventory 2020 & 2013	Agriculture and Agri-food Canada. (2020). *Annual Crop Inventory—Open Government Portal* (Version 1) [.shp]. https://open.canada.ca/data/en/dataset/ba2645d5-4458-414d-b196-6303ac06c1c9 Agriculture and Agri-food Canada. (2013). *Annual Crop Inventory—Open Government Portal* (Version 1) [.shp]. https://open.canada.ca/data/en/dataset/ba2645d5-4458-414d-b196-6303ac06c1c9
Pre-2000s agriculture – Land Cover for Agricultural Regions of Canada	Agriculture and Agri-food Canada. (2009). *Land Cover for Agricultural Regions of Canada, circa 2000—Open Government Portal* (Version 1) [Raster]. https://open.canada.ca/data/en/dataset/16d2f828-96bb-468d-9b7d-1307c81e17b8?_ga=2.1988981.515457352.1493845476-1970616929.1493845475
Open Database of Buildings	Statistics Canada, G. of C. (2019). *The Open Database of Buildings* [.shp]. https://www.statcan.gc.ca/en/lode/databases/odb
Alaska hydrographic features - 100k	U.S. Geological Survey. (2022). *National Hydrography Dataset (NHD). State—Alaska* [.shp]. https://www.sciencebase.gov/catalog/item/5136012ce4b03b8ec4025bf7
Montana hydrographic features -24k	U.S. Geological Survey, & Montana State Library. (2003). *Montana Hydrography Framework (National Hydrography Dataset)* [.shp]. https://mslservices.mt.gov/Geographic_Information/Data/DataList/datalist_Details.aspx?did=%7bdb6c41bd-1f29-48ab-b4aa-2c1890f317e6%7d
Provincial – 2000s	Natural Resources Canada. (2014). Topographic Data of Canada—CanVec 1:50,000, 1944-2013—Open Government Portal [.shp]. https://open.canada.ca/data/en/dataset/be0165a8-ad5d-4adb-a27a-2d4117c3967c

6.3.2 CanMatrix maps for time-step 1970-1999

Natural Resources Canada. (2014). *Digital Topographic Raster Maps—Archived* (Version 4) [Raster]. https://ftp.maps.canada.ca/pub/nrcan_rncan/raster/topographic

For full list of maps per park-control pair, please see associated spreadsheet.

6.3.3 Topographic NTS maps for time-step 1931-1969

Army Survey Establishment, G. of C., & Borealis. (2022). *Historical National Topographic System (NTS): 1:50,000 Scale Maps, Data and GIS* [Raster]. https://borealisdata.ca/dataverse/topomaps

Army Survey Establishment, G. of C., & Natural Resources Canada. (2020). Historical Topographic Maps: 1:50,000 Index [Raster]. https://geo2.scholarsportal.info

For full list of maps per park-control pair, please see associated spreadsheet.

6.3.4 Sectional Maps collection at the University of Calgary (SANDS) for time-step pre-1930

SANDS, U. of C. (n.d.). *Sectional Maps, 1 to 3 Mile. 1902 to 1955* [Raster]. https://sands.ucalgary.ca/App/SectionalMaps/index.html

6.3.5 Other maps used from external collections

Name	Year	Scale	Source	Collection
Waterton Lakes National Park	1914	1:62,500	Office of the Surveyor General	Historical Maps Collection, Libraries and Cultural Resources Digital Collections, University of Calgary
Crowsnest Forest and Waterton Lakes Park sheet 4	1914	1:62,5000	Office of the Surveyor General	Historical Maps Collection, Libraries and Cultural Resources Digital Collections, University of Calgary
Crowsnest Forest and Waterton Lakes Park sheet 5	1914	1:62,5000	Office of the Surveyor General	Historical Maps Collection, Libraries and Cultural Resources Digital Collections, University of Calgary
Yoho Park	1921	1:125,000	Department of the Interior	University of Alberta William C. Wonders Map collection
Mount Revelstoke Park	1923	1:125,000	Department of the Interior	UBC Koerner Library
Prince Albert National Park	1923	na	Department of the Interior	Canadiana Heritage Reel T-10406
Jasper National Park	1926	na	Department of the Interior, Engineering Service Canadian National Parks	UBC Koerner Library
Kootenay National Park	1926	1:126,720	Department of the Interior	Canadiana Heritage Reel T-12420
Pelee	1926	1:63,360	Department of National Defence, Geographical Section, General Staff	McMaster University Map collections - Ontario Historical Topographic Maps
Waterton Lakes Park	1928	1:63,360	Department of the Interior	UBC Koerner Library
Wood Buffalo Park	1931	1:506,880	Department of the Interior	U of T Libraries scanned maps
Kootenay National Park	1938	1:126,720	Mines and Technical Surveys. Surveys and mapping branch	Topographic map cabinets
Waterton Lakes	1940	1:63,360	Mines and Technical Surveys. Surveys and mapping branch	University of Alberta William C. Wonders Map collection

Wood Buffalo Park	1947	1:506,880	Department of Mines and Resources. Surveys and Engineering Branch.	U of T Libraries scanned maps
Prince Albert Park	1951	1:150,000	Mines and Technical Surveys. Surveys and mapping branch	University of Alberta William C. Wonders Map collection
Riding Mountain Park	1954	1:190,080	Surveys and Mapping Branch. Energy, Mines and Resources Canada	UBC Koerner Library
Banff National Park	1955	1:190,080	Mines and Technical Surveys. Surveys and mapping branch	National Park Files – Concordia University
Waterton Lakes	1955	1:50,000	Mines and Technical Surveys. Surveys and mapping branch	www.canadiana.ca
Waterton Lakes Park	1958	1:63,360	Mines and Technical Surveys. Surveys and mapping branch	UBC Koerner Library
Sage Creek	1960	1:50,000	Mines and Technical Surveys. Surveys and mapping branch	www.canadiana.ca
Beaver Mines AB/BC	1960	1:50,000	Mines and Technical Surveys. Surveys and mapping branch	www.canadiana.ca
Beaver Mines AB	1960	1:50,000	Mines and Technical Surveys. Surveys and mapping branch	www.canadiana.ca
Yoho Park	1961	1:126,720	Mines and Technical Surveys. Surveys and mapping branch	National Park Files – Concordia University
Mount Revelstoke Park	1963	1:50,000	Department of Mines and Resources. Surveys and Engineering Branch.	National Park Files – Concordia University
Blairmore AB	1967	1:50,000	Surveys and Mapping Branch. Energy, Mines and Resources Canada	www.canadiana.ca
Pelee Point	1969	1:25,000	Surveys and Mapping Branch. Energy, Mines and Resources Canada	McMaster University Map collections - Ontario Historical Topographic Maps

6.4 R packages

Ben-Shachar, M. S., Makowski, D., Lüdecke, D.,, I., Wiernik, B. M., Thériault, R., Kelley, K., Stanley, D., Caldwell, A., Burnett, J., Karreth, J., & Waggoner, P. (2024). *effectsize: Indices of Effect Size* (0.8.9) [Computer software]. https://cran.r-project.org/web/packages/effectsize/index.html

Elzhov, T. V., Mullen, K. M., Spiess, A.-N., & Bolker, B. (2023). *minpack.lm: R Interface to the Levenberg-Marquardt Nonlinear Least-Squares Algorithm Found in MINPACK, Plus Support for Bounds* (1.2-4) [Computer software]. https://cran.r-project.org/web/packages/minpack.lm/index.html

Grothendieck, G., & Team (nls), R. C. (2024). *nls2: Non-Linear Regression with Brute Force* (0.3-4) [Computer software]. https://cran.r-project.org/web/packages/nls2/index.html

Mazerolle, M. J. (2023). *AICcmodavg: Model Selection and Multimodel Inference Based on (Q)AIC(c)* (2.3-3) [Computer software]. https://cran.r-project.org/web/packages/AICcmodavg/index.html

6.5 R code

```
# Load packages
require(minpack.lm) # Fitting non-linear models
require(nls2) # Fitting non-linear models
require(AICcmodavg) # calculate second order AIC (AICc)
require(effectsize)


#set wd to folder with data
setwd()


### Create the ProgressiveChangeBACIPS function
ProgressiveChangeBACIPS<-function(control, impact, time.true, time.model)
{
  ### STEP 2 - Calculate the delta at each sampling date
  delta <- impact - control
  print(delta)

  # Plot delta against time.true ORIGINAL
  dev.new(width=10, height=5)
  par(mfrow=c(1,2))
  plot(delta~time.true, type="n")
  time.model.of.impact=max(which(time.model==0))
  rect(time.model.of.impact-100, min(delta)-100, time.model.of.impact+1, max(delta)+100, col = "grey")
  points(delta~time.true, pch=24, bg="white", cex=2)

  ### STEP 3 - Fit and compete models
  ## Create a 'period' variable
  period <- ifelse(time.model==0, "Before","After")

  ## Fit a step model
  step.Model<-aov(delta ~ period)

  ## Fit a linear model
  linear.Model<-lm(delta ~ time.model)

  ## Fit an asymptotic model
  # Create an asymptotic function
  myASYfun<-function(delta, time.model)
  {
    funAsy<-function(parS, time.model)  (parS$M * time.model) / (parS$L + time.model) + parS$B
    residFun<-function(p, observed, time.model) observed + funAsy(p,time.model)
    parStart <- list(M=mean(delta[time.model.of.impact:length(time.true)]), B=mean(delta[1:time.model.of.impact]), L=1)
    nls_ASY_out <- nls.lm(par=parStart, fn= residFun, observed=delta, time.model=time.model, control = nls.lm.control(maxfev = integer(), maxiter = 1000))
    foAsy<-delta~(M * time.model) / (L + time.model) + B
    startPar<-c(-coef(nls_ASY_out)[1], coef(nls_ASY_out)[2], coef(nls_ASY_out)[3])
    asyFit<-nls2(foAsy, start=startPar, algorithm="brute-force") # nls2 enables to calculate AICc
    asyFit
  }
  # Fit the asymptotic model
  asymptotic.Model<-myASYfun(delta=delta,time.model=time.model)
```

```
## Fit a sigmoid model
## Create a sigmoid function
mySIGfun<-function(delta, time.model)
{
  funSIG<-function(parS, time.model)  (parS$M * (time.model/parS$L)^parS$K) / (1 + (time.model/parS$L) ^ parS$K) + parS$B
  residFun<-function(p, observed, time.model) observed + funSIG(p,time.model)
  parStart <- list(M=mean(delta[time.model.of.impact:length(time.true)]), B=mean(delta[1:time.model.of.impact]), L=mean(time.model), K=5)
  nls_SIG_out <- nls.lm(par=parStart, fn= residFun, observed=delta, time.model=time.model, control=nls.lm.control(maxfev = integer(), maxiter = 1000))
  foSIG<-delta~(M * (time.model/L) ^ K) / (1 + (time.model/L) ^ K) + B
  startPar<-c(-coef(nls_SIG_out)[1],-coef(nls_SIG_out)[2],coef(nls_SIG_out)[3],coef(nls_SIG_out)[4])
  sigFit<-nls2(foSIG, start=startPar, algorithm="brute-force") # nls2 enables to calculate AICc
  sigFit
}
# Fit the sigmoid model
sigmoid.Model<-mySIGfun(delta=delta,time.model=time.model)


## Compare models
# Perform AIC tests
AIC.test<-AIC(step.Model, linear.Model, asymptotic.Model, sigmoid.Model)
AICc.test<-as.data.frame(cbind(AIC.test[,1], c(AICc(step.Model), AICc(linear.Model), AICc(asymptotic.Model), AICc(sigmoid.Model))))
rownames(AICc.test)<-rownames(AIC.test)
names(AICc.test)<-names(AIC.test)

# Calculate AICc weight and select the best model
for(i in 1:dim(AICc.test)[1])
{
  AICc.test$diff[i]<-AICc.test$AIC[i]-min(AICc.test$AIC)
}
AICc.test$RL<-exp(-0.5* AICc.test$diff)
RL_sum<-sum(AICc.test$RL)
AICc.test$aicWeights<-(AICc.test$RL/RL_sum)*100
w<-AICc.test$aicWeights
names(w)<-rownames(AICc.test)

# Display AICc weights
print(w)
barplot(w, col="white", ylab="Relative likelihood (%)", cex.names = 0.9, names.arg =c("Step","Linear","Asymptotic","Sigmoid"))
best.Model<-which(w==max(w))

#example: Taiga region
# import dataframe
  Site<-read.csv("Taiga.csv")

# attach dataframe DON'T FORGET
attach(Site)

#Test assumptions
#Uses graphical techniques due to small sample sizes for individual parks.
summary(Site)

#Normality
par(mfrow=c(1,2))
qqnorm(control)
qqline(control)

qqnorm(impact)
qqline(impact)

#If p-value <= 0.05, data is likely normally distributed.
#DO NOT USE FOR SMALL SAMPLE SIZES. Exploratory graphical techniques best here
shapiro.test(control)
shapiro.test(impact)

#Homoscedasticity. #if box widths are equal, it supports the assumption of homoscedasticity.
boxplot(control)
boxplot(impact)

#Autocorrelation
acf(control)
acf(impact)
#If the values are within confidence intervals, suggests that the autocorrelation is not statistically significant.

# Run ProgressiveChangeBacips function
ProgressiveChangeBACIPS(control, impact, time.true, time.model)

detach(Site)
```