INTRODUCTION

Trout and charr, members of the salmonid family, have high conservation value owing to their cultural, economic and ecological importance (Behnke, 2002; Brodie, Redford, & Doak, 2018; Hammerschlag et al., 2019; Holmlund & Hammer, 1999; Prosek, 2013). Many trout and char species support commercial, recreational and/or subsistence fisheries with cultural and economic benefits (Holmlund & Hammer, 1999; Liu, Bailey, & Davidsen, 2019; Zeller, Booth, Pakhomov, Swartz, & Pauly, 2011). Requiring cold, clean waters, salmonids are often used as indicator species for assessing ecosystem health (Crisp, 2000). Salmonids have complex life-histories that require access to multiple discrete habitat types (e.g. headwater streams, large rivers, lakes, estuaries, oceans) to meet various needs throughout life (Behnke, 2002) and as such, their presence is a reflection of diverse habitats and connectivity.
They play important roles in ecosystem function by asserting top-down control as aquatic predators (Hammerschlag et al., 2019), moving nutrients between habitats (Gende, Edwards, Willson, & Wipfli, 2002) and providing food for terrestrial wildlife (Hilderbrand, Farley, Schwartz, & Robbins, 2004). While their complex life-history lets them act as a conduit among ecosystems, this characteristic also makes them susceptible to anthropogenic threats because of the specificity of their habitat requirements (Behnke, 2002). A recent review by Muhlfeld et al. (2018) highlighted the widespread conservation concern for “trout,” reporting that 73% of those species assessed by the International Union for Conservation of Nature (IUCN) are currently at risk (i.e. vulnerable, endangered, critically endangered) or extinct. Muhlfeld and coauthors stressed the need to address current and future threats to these species to promote their recovery.

In Canada, species with conservation concerns are assessed by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), an independent advisory panel to the Minister of Environment and Climate Change Canada. Available data and information are compiled into status reports which COSEWIC uses to determine the at-risk designation of the species or relevant subspecies units. Designatable units (DUs) are groups below species level used by COSEWIC to represent a subspecies or population which is discrete and evolutionarily significant (COSEWIC, 2017a). The designation categories used by COSEWIC in order of increasing severity are not at risk, special concern, threatened, extirpated and extinct (COSEWIC, 2017b). If designated at risk at any level by COSEWIC, the DU is recommended for listing under the Canadian Species at Risk Act (SARA) by the Minister of Environment and Climate Change of Canada.

Of freshwater trout and charr in the Canadian Rocky Mountain region, westslope cutthroat trout (*Oncorhynchus clarkii lewisi*), bull trout (*Salvelinus confluentus*; a char species) and Athabasca rainbow trout (*Oncorhynchus mykiss*) are of conservation concern and have been assessed by COSEWIC (2012, 2014, 2016; Figure 1; Table 1). Both westslope cutthroat trout DUs located within the Canadian Rocky Mountain region were assessed in 2005: Saskatchewan-Nelson Rivers populations—threatened; and Pacific populations—special concern; and were listed under SARA in 2006. Three bull trout DUs are located within Canadian Rocky Mountain watersheds and were assessed in 2012: Western Arctic populations—special concern; Saskatchewan-Nelson Rivers populations—threatened; and Pacific populations—not at risk. Both at-risk DUs were listed under SARA in 2019 (The Canada Gazette, 2019). The Athabasca River population of rainbow trout, hereafter called Athabasca rainbow trout, was assessed as endangered in 2014 and listed under SARA in 2019 (The Canada Gazette, 2019).

While COSEWIC assessments provide a summary of population statuses (i.e. whether population densities are diminished or declining) and threats as understood at the DU level, they do not compare statuses or threats among regions or species at an ecosystem level. Examining conservation needs for multiple species together will allow prioritisation of conservation efforts when resources are limiting and will allow us to examine landscape patterns in threats while also considering species-specific susceptibility determined by biological characteristics. Here, we compared threats facing the three native at-risk trout and charr in watersheds supported by the Canadian Rocky Mountains: two westslope cutthroat trout DUs, two bull trout DUs and one rainbow trout DU (Figure 1). To achieve this comparison, we assessed the relative severity of historical threats within and among species by conducting a comprehensive review of the qualitative threats discussed in existing COSEWIC assessment reports and provincial reports. We also assessed the relative severity of future threat impacts by comparing IUCN threat assessments conducted by COSEWIC.

## 2 SPECIES DISTRIBUTION AND DENSITY

In addition to the provinces of British Columbia (BC) and Alberta, and Yukon and Northwest Territories (NWT), several national parks are included in the range of the DUs of interest (i.e. Jasper, Yoho, Kootenay, Banff, Glacier, Nahanni and Waterton Lakes National Parks). Fish populations within the national parks are managed by Parks Canada according to the National Parks Act, and those outside of national parks are managed according to provincial or territorial regulations. COSEWIC’s assessment of risk considers the Extent of Occurrence, which is the area encompassing the range of all known populations of the DU, and the Index of Area of Occupancy, which reflects the area within its Extent of Occurrence that is occupied by the species or DU. COSEWIC uses these metrics as part of its quantitative assessment to determine at-risk level. We present these metrics below to provide a sense of the geographic extent of each DU.

### 2.1 Westslope cutthroat trout: Pacific populations

The native range of westslope cutthroat trout in BC is concentrated on the western slope of the Rocky Mountains in the south-eastern part of the province and is contained within four watersheds that make up the core range (i.e. Elk, Flathead, Upper Kootenay, West Kootenay), and three watersheds that make up the peripheral range where occurrence is more sparse (i.e. Columbia, Kettle, South Thompson; Figure 1; BC MOE, 2014). The Pacific DU of westslope cutthroat trout has an Extent of Occurrence of 85,183 km² and an Index of Area of Occupancy of 6,824 km² (Table 1). Here, westslope cutthroat trout are thought to be present throughout their historical range and are found in an estimated 1,319 water bodies, 928 of which have no records of historical stocking. Our understanding of the current distribution of native westslope cutthroat trout in BC is confounded by a history of stocking of westslope cutthroat trout, which introduced westslope cutthroat trout outside their native range, and rainbow trout, which created opportunities for rainbow trout and westslope cutthroat trout to hybridise (BC MOE, 2014).
Density data are sparse for this DU but have been collected for a few high priority streams in the Upper Kootenay and Elk regions (BC MOE, 2014). Population statuses of BC westslope cutthroat trout are determined by comparing electrofishing catch per unit effort (CPUE) to a benchmark density from large productive catch-and-release rivers (Hagen & Baxter, 2009). Of the 10 river reaches...
assessed, six reaches showed stable, productive or recovering densities compared to the benchmark density (BC MOE, 2014). The two population trends known in this DU show a general increase in CPUE (BC MOE, 2014).

2.2 | Westslope cutthroat trout Saskatchewan-Nelson Rivers populations

The native range of westslope cutthroat trout in Alberta is concentrated in the south-western part of the province along the east slopes of the Rocky Mountains mainly in the Bow and Oldman river basins (Figure 1). It has been substantially reduced from its historical range with contraction evident on the eastern part of its range (Figure 2a). Only 51 genetically pure subpopulations remain out of the original 274 estimated in Alberta (excluding the national parks), and 71% of its originally occupied watershed units have been extirpated (Figure 2a; AEP, 2019a; COSEWIC, 2016). The Extent of Occurrence and Index of Area of Occupancy were estimated at 16,650 km² and 844 km², respectively (COSEWIC, 2016).

For Alberta watersheds, historical adult density metrics provide context with respect to pre-impact densities using information from several sources (e.g. test netting, seine netting and electrofishing CPUE; test angling; historical and traditional knowledge; published and unpublished “grey” literature; and professional opinion). Historical populations contained predominantly high or very high densities of westslope cutthroat trout, particularly in the headwater and foothill regions (Figure 2a; AEP, 2019a). Similar to BC’s westslope cutthroat trout population assessments, population status in Alberta jurisdictions are also determined by comparison to benchmark densities from sites thought to represent undisturbed populations within the region (MacPherson, Coombs, Reilly, Sullivan, & Park, 2014). Since it is difficult to find an appropriate undisturbed westslope cutthroat trout population to act as a reference, the benchmark density value was based on the lowest densities observed over the past 20 years at the top five westslope cutthroat trout streams and rivers (pers. comm. L. MacPherson). Compared to these benchmark densities, most of the Alberta subpopulations currently have very low densities (26%) or are functionally extirpated (71%; Figure 2a; AEP, 2019a). Recent rates of decline in the remaining populations are not explicitly known (COSEWIC, 2016).

2.3 | Bull trout: Saskatchewan-Nelson Rivers populations

Bull trout in the Saskatchewan-Nelson Rivers populations DU is contained entirely within Alberta and comprises the North and South Saskatchewan River basins in the south-western part of the province (Figure 1; COSEWIC, 2012). The distribution of this DU has...
contracted somewhat from its historical range which once included populations further downstream along the major river networks into the parkland and prairie regions (Figure 2b; AEP, 2019a; ASRD, 2012). Seven of the 48 watersheds originally in this DU are now extirpated, all of which were located at the downstream extent of the range. The Extent of Occurrence and Index of Area of Occupancy were estimated at > 20,000 km² and > 2,000 km², respectively, (Table 1; COSEWIC, 2012).

Historical populations were mostly characterised by high and very high densities (Figure 2b; AEP, 2019a). To determine current population statuses, density estimates were compared to reference sites within the DU that were selected owing to their high catch rates and minimal anthropogenic impacts (pers. comm. J. Reilly). Most of the watershed units within this DU currently have low (16%) or very low densities (42%), or are considered functionally extirpated (16%; Figure 2b). The few remaining watershed units with moderate or high densities are in the headwaters in the northern part of the region. An assessment of short-term trends conducted by provincial fisheries staff identified the most severely declining populations within the central watersheds in this DU (i.e. Red Deer and Bow River watersheds), and the most stable populations within the northern-most watersheds in the DU (i.e. North Saskatchewan watershed; ASRD & ACA, 2009a).

### 2.4 | Bull trout: Western Arctic populations

The Western Arctic populations of bull trout are constrained to the MacKenzie River drainage including the Liard, Peace and Athabasca River basins (Figure 1). It is located partly in Alberta, extending north-west into northern BC, NWT and a small part of Yukon Territory. The Extent of Occurrence and Index of Area of Occupancy are > 20,000 km² and > 2,000 km², respectively (Table 1, COSEWIC, 2012). The range of this DU has seen some contraction from its historical expanse in the Alberta section (Figure 2b), but, whether the current distribution differs from historical in BC, NWT or Yukon is not known owing to a lack of historical information.

Historical records in the Alberta portion of the DU indicate that bull trout existed in high and very high densities in the headwater and foothill areas but in moderate and low densities further from the mountains (Figure 2b; AEP, 2019a). The Alberta portion of the Western Arctic populations DU uses the same reference populations as the Saskatchewan-Nelson Rivers populations DU to determine current population status (pers. comm. J. Reilly). Most Alberta watershed units in this DU currently have low (23%) or very low (28%) densities, or are functionally extirpated (28%); a few healthier populations remain in the headwater regions. An assessment of short-term trends by Alberta fisheries staff indicated the greatest decline in the Athabasca River basin, located at the southern end of this DU (ASRD & ACA, 2009a). Interestingly, the Peace River population in Alberta (at the far north of the Alberta distribution) was not present in historical records but its current presence, though in very low densities, is thought to result from increased cold-water habitat in Alberta following upstream dam construction (pers. comm. K. Wilcox cited in AEP, 2019a). BC bull trout status assessments use methods developed by the US Fish and Wildlife Service (USFWS), which combine adult abundances, distribution and trends with threats for a given area (Hagen & Decker, 2011; USFWS, 2005). Abundance data are derived from redd counts, migrating adult counts, and aerial, snorkeling and creel surveys. Abundance, distribution, population trends and threats are scored following USFWS ranges and are not specific to BC (Hagen & Decker, 2011; USFWS, 2005). Current abundance data on the BC portion of the DU is sparse. At the time of a provincial assessment of bull trout in BC, Hagen and Decker (2011) reported that abundance was unknown for 25 of 30 core areas and only five core areas could be evaluated using expert opinion. Of those five core areas, three were identified as at-risk and two were potential risk (Hagen & Decker, 2011). Even less is known about the number and size of populations in the Yukon and Northwest Territories. Population estimates that have been conducted generally reflect small populations which have been attributed to low productivity (COSEWIC, 2012).

### 2.5 | Rainbow trout: Athabasca River populations

While native rainbow trout are widely distributed in western Canada, they are only present in three river basins east of the continental divide: Liard, Peace and Athabasca. The Athabasca population is considered a DU satisfying the discreteness and evolutionary significance criteria outlined by COSEWIC (2014, 2017a; Figure 1). Athabasca rainbow trout are considered extirpated from Jasper National Park as a result of substantial introgression with stocked non-native rainbow trout and are no longer considered part of the DU (Figure 2c; AEP, 2019a; COSEWIC, 2014). Outside of Jasper National Park, the historical range has remained intact. The Extent of Occurrence and Index of Area of Occupancy were estimated at 24,450 and 2,560 km², respectively (Table 1; COSEWIC, 2014).

Historical population densities were variable, ranging from very low to very high densities, with the highest densities in watersheds immediately downstream of Jasper National Park. Current population statuses were determined by comparing density estimates to benchmark densities established at unharvested streams in a reference watershed (Tri-Creeks experimental watershed; ASRD & ACA, 2009b). Currently, almost all watersheds have very low population densities (Figure 2c). Temporal assessment of abundance data showed declines over 15 recent years in approximately 54% of streams where appropriate data exist (COSEWIC, 2014). The best estimate of the rate of decline in the abundance of Athabasca rainbow trout for the whole DU was estimated at approximately 96.5% based on time series over 52 sites.

### 3 | BIOLOGICAL AND ECOLOGICAL CHARACTERISTICS

The historical range of a fish species in the Canadian Rocky Mountain region is the result of its post-glacial distribution and dispersal
(Hocutt & Wiley, 1986) and how natural variation in climate and habitat characteristics have interacted with its biological and ecological requirements. A species’ current distribution additionally reflects its response to historical anthropogenic threats and depends, in part, on the unique biological and ecological characteristics of the species and/or life-history form.

Westslope cutthroat trout and bull trout each have three life-history forms in the Canadian Rocky Mountain region: stream-resident, fluvial and adfluvial (Table 2; ASRD & ACA, 2009a; COSEWIC, 2016). The stream-resident form of both species completes its entire life cycle in small headwater streams and reaches the smallest maximum lengths with fork lengths below 300 mm (reviewed in ASRD & ACA, 2009a; COSEWIC, 2016). The fluvial and adfluvial forms are both migratory and adults migrate from their overwintering grounds in larger rivers (fluvial) or lakes (adfluvial) to spawn in small headwater streams. For both migratory forms, juvenile rearing also occurs within small headwater streams (reviewed in ASRD & ACA, 2009a; COSEWIC, 2016). Both migratory forms of westslope cutthroat trout are known to reach fork lengths of over 500 mm (reviewed in COSEWIC, 2016). The migratory forms of bull trout are also larger than their stream-resident counterparts with the fluvial form reaching 400 mm on average, and the adfluvial form being the largest of the three (reviewed in ASRD & ACA, 2009a). In contrast, 90% of Athabasca rainbow trout are stream residents (fork length < 300 mm), with only 10% migrating downstream to higher order rivers (fork length can reach over 400 mm, Sterling, 1990).

Bull trout are the longest lived of the three species, with maximum ages ranging from approximately 10 to 20 years (Table 2). They exhibit variation among life-history forms with the shortest longevity observed in stream-resident fish, and the longest longevity observed in adfluvial fish (ASRD, 2012). Athabasca rainbow trout and westslope cutthroat trout are shorter lived with few individuals living past 8 and 5 years, respectively (Sterling, 1990; A. Costello per. comm. in ASRD & ACA, 2006). Age-at-maturity is similar for westslope cutthroat trout and Athabasca rainbow trout (most individuals mature by 4–5 years), whereas most bull trout mature between 5 and 7 years old (Table 2; ASRD & ACA, 2009a; COSEWIC, 2014; Nelson & Paetz, 1992; Scott & Crossman, 1973). All three species are iteroparous, although westslope cutthroat trout and bull trout have been known to spawn in alternate years (DFo, 2018; Johnston & Post, 2009; The Alberta Westslope Cutthroat Trout Recovery Team, 2013). Given the short life-span of westslope cutthroat trout (most ≤ 5 years), few repeat spawners exist despite their iteroparity (ASRD & ACA, 2006); this likely also applies to Athabasca rainbow trout to some degree. Bull trout is the only annual spawner of the group, spawning between mid-August and late October when temperatures approach 7 to 15°C (reviewed in Sinnatamby et al., 2018). Westslope cutthroat trout spawn between May and July, typically when temperatures increase to 10°C (Nelson & Paetz, 1992) or as low as 6°C in cold, high elevation lakes (S. Humphries pers. comm. in COSEWIC, 2016). Athabasca rainbow trout spawn between late May and early June following peak flows from the spring snowmelt once daily maximum water temperatures reach approximately 6°C (Sterling, 1992). They tend to spawn later and in finer substrate compared to non-native rainbow trout (Sterling, 1990).

While all three species are opportunistic, generalist feeders, bull trout display increasing piscivory with size when forage fish are available (Table 2; COSEWIC, 2012, 2014, 2016). On the other hand, westslope cutthroat trout have shown a preference for invertebrates even when forage fish are abundant (Shepard, Pratt, & Graham, 1984) and Athabasca rainbow trout also feed primarily on aquatic and terrestrial insects (The Alberta Athabasca Rainbow Trout Recovery Team, 2014).

All three species require cold, clean streams with abundant cover and access to groundwater seeps for overwintering (Table 2; COSEWIC, 2012, 2014, 2016). Migratory forms of each species require connectivity to access discrete habitats at different life stages. All three species prefer cold water and have similar thermal preferences (Table 2); however, Athabasca rainbow trout have the highest tolerance for warmer waters evidenced by the highest upper incipient lethal (UILT) and optimal incubation temperatures (Bear, McMahon, & Zale, 2007; Sterling, 1986). While westslope cutthroat trout and bull trout have similar ULTs and optimal temperatures for growth, bull trout require much colder incubation temperatures suggesting a narrower requirement (Berry, 1994; Bear et al., 2007; Fairless, Herman, & Rhem, 1994; Merriman, 1935; Selong, McMahon, Zale, & Barrows, 2001). All three species spawn in small streams, with females building redds in well-oxygenated gravel substrates (Baxter & Hauer, 2011; Brown & Mackay, 2008; Sterling, 1990).

## 4 | HISTORICAL AND FUTURE THREATS

The current distribution and densities of the three at-risk trout and char species in the Canadian Rocky Mountain region reflect their historical distribution, and the interaction between biological/ ecological characteristics and landscape patterns in the presence and severity of historical threats. Historical threats within the Canadian Rocky Mountain region range of westslope cutthroat trout, bull trout and Athabasca rainbow trout were compared using qualitative discussions presented in existing reports (ASRD & ACA, 2006, 2009a, 2009b; COSEWIC, 2012, 2014, 2016).

In 2012, COSEWIC approved a motion to include a future threat assessment in all future status reports. The future threat assessment is conducted by a panel of experts using the IUCN threats calculator. The expert panel considers ongoing and future threats in currently inhabited areas (i.e. excluding extirpated populations) and generates threat impact scores of negligible, low, medium, high or very high for each individual threat category. Where uncertainty exists, the expert panel can assign a range of threat levels (e.g. medium-high). Using an algorithm, scores for the individual threat categories comprising scope, severity and timing of the threat are combined to produce a calculated overall threat impact for each DU. The calculated overall threat impact provides general guidance but is reviewed and
can be adjusted by the expert panel to produce a final assigned overall threat impact (Faber-Langendoen et al., 2012). An IUCN threat assessment was completed and included in the COSEWIC reports for Athabasca rainbow trout (2014) and westslope cutthroat trout (2016) but was not completed for bull trout (2012) since its assessment was done prior to this procedural change. To facilitate a comparison of ongoing and future threats facing all three species, we gathered a panel of regional experts on bull trout to conduct a threat assessment using the IUCN threats calculator. Owing to the comparative wealth of information that exists for the Saskatchewan-Nelson Rivers populations, our threat assessment focused on this DU. A future threat assessment was not completed for the Western

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Arctic populations DU that encompasses a vast geographic area under four provincial and territorial jurisdictions and has relatively sparse data. Both historical and future threats can be considered in four broad categories: angling, non-native species and genes, habitat loss and alteration, and climate change. In the following subsections, we present the results of our qualitative assessment, comparing historical and future threat severity across DUs.

4.1 | Angling

Arguably, the largest historical threat for both species in the south-western region of Alberta (i.e. westslope cutthroat trout and bull trout Saskatchewan-Nelson Rivers populations DUs) was overexploitation from angling. Both species were targeted by anglers in the 19th and 20th centuries but for different reasons. Bull trout were regarded poorly by anglers and were harvested with vigour in an effort to eradicate them and make room for preferred species (Colpitts, 1997). Westslope cutthroat trout were, and remain, a popular recreational angling target (Colpitts, 1997; Mayhood, 2009; Prince, 1912). Declines in westslope cutthroat trout numbers were observed coincident with the arrival of the Canadian Pacific Railway in the late 19th century when native salmonids were harvested in large numbers (Mayhood, 2009). Both species are considered easy to catch because of their voracious feeding habits and accessibility in small streams or when congregate in spawning migrations (MacPhee, 1966; Post, Mushens, Paul, & Sullivan, 2003; Prince, 1912). The historical impacts of angling on westslope cutthroat trout and bull trout are less apparent in the other regions (i.e. Pacific, Western Arctic) likely owing to a combination of factors including reduced accessibility, lower human population densities, and/or higher natural fish densities. Comparatively, historical fishing on Athabasca rainbow trout was seemingly less intense than that experienced by native trout in the south-west of the province, but fishing was suggested as a historical threat based on high densities in two streams that have been closed to fishing since the 1960s. A third closed stream with notably lower densities was attributed to naturally ill-suited habitat, so the relative roles of fishing and suitable habitat could not be determined (ASRD & ACA, 2009b).

The ongoing and future threat posed by angling to each DU, as estimated by the IUCN threats calculator, is considerably lower than historical levels owing to contemporary restrictive fishing regulations: a change to catch-and-release regulations throughout Alberta for bull trout (in 1995), Athabasca rainbow trout outside of Jasper National Park (in 2012) and pure populations of westslope cutthroat trout (in 2016); BC fishing regulations on westslope cutthroat trout have become more conservative since the 1980s, with restrictive size limits on harvest (Figure 3). Restrictive regulations have resulted in recovery of some populations (BC MOE, 2014; Herman, 1997; Johnston et al., 2007; Parker, Schindler, Wilhelm, & Donald, 2007) but no consistent region-wide response has been observed. Despite restrictive regulations, angling still poses a threat due to poaching, accidental misidentification and post-release hooking mortality. The percentage of fish that die post-release is unknown but has been estimated at 8% (median value from a meta-analysis) and can vary with species, angler skill, type of bait, type of gear, number of hooking events and water temperature (Bartholomew & Bohnsack, 2005). Bull trout are likely the most susceptible to hooking mortality because of their high catchability and sensitivity to high temperatures (Prince, 1912; Selong et al., 2001).

4.2 | Non-native species and genes

Government agencies began introducing non-native trout (e.g. rainbow trout, brook trout (Salvelinus fontinalis), brown trout (Salmo trutta), various cutthroat trout subspecies and golden trout (Oncorhynchus aguabonita)) throughout BC and Alberta in the early 20th century to satisfy the recreational fishing preferences of newcomers from eastern Canada, New England States and Europe (Colpitts, 1997). Many of these species have become naturalised and continue to live in the ranges occupied or previously occupied by native trout (Paul & Post, 2001). Lake trout (Salvelinus namaycush) are native to the Canadian Rocky Mountain region but have been introduced into systems where they were not naturally found and are also included in this section (Donald & Alger, 2008). Non-native trout can impact native trout populations, in part, through competition and/or hybridisation (Gozlan, Britton, Cowx, & Copp, 2010). Of all the DUs compared, the historical severity of the non-native species threat was highest for westslope cutthroat trout in Alberta (Saskatchewan-Nelson Rivers populations DU). Here, westslope cutthroat trout faced a number of introduced trout species (i.e. non-native rainbow trout, Yellowstone cutthroat trout (Oncorhynchus clarkii bouvieri), golden trout, brook trout, brown trout and lake trout; COSEWIC, 2016) but by far, the largest historical threat was from non-native rainbow trout, which can both compete for resources and hybridise with westslope cutthroat trout (COSEWIC, 2016). As a result, numerous cutthroat trout populations still exist on the landscape but only a small number are considered genetically pure (COSEWIC, 2006, 2016; AEP, 2019b). The historical threat of hybridisation appears less severe in the Pacific populations DU, with only two of four core regions showing significant amounts of introgression, although, the genetic integrity of many populations in this DU remains unknown (BC MOE, 2014). Brook trout was also a large historical threat to westslope cutthroat trout, potentially displacing whole populations via competition in areas where brook trout were stocked (The Alberta Westslope Cutthroat Trout Recovery Team, 2013). Notably, the adfluvial form of westslope cutthroat trout has been entirely replaced by brook trout in national parks where they were stocked in BC (BC MOE, 2014).

Similarly, brook trout and non-native rainbow trout were the main historical non-native species threats to Athabasca rainbow
trout. Non-native hatchery rainbow trout were stocked into the Athabasca watershed in Jasper National Park in 1919 (DFO, 2018) and outside of the park in 1955 (AEP, 2019b). Hybridisation has resulted in low genetic purity (admixture coefficient ($Q_i$) < 0.85) in much of Jasper National Park and consequently, COSEWIC no longer includes rainbow trout within Jasper National Park as native members of the DU (COSEWIC, 2014). Outside of Jasper National Park, populations of Athabasca rainbow trout have remained mostly pure ($Q_i > 0.95$) in some streams with a history of stocking of non-native rainbow trout (COSEWIC, 2014), suggesting a less severe historical impact of non-native rainbow trout here relative to that observed in south-west Alberta on westslope cutthroat trout. Brook trout were stocked into lakes and some creeks in Jasper National Park between 1924 and 1977, and into streams in the Athabasca watershed outside of Jasper National Park between 1940 and 1964 (COSEWIC, 2014). Since then brook trout have increased in many of the stocked streams and expanded into some of the non-stocked streams as well, now comprising as much as 100% of the trout population in some streams (ASRD & ACA, 2009b). The Athabasca watershed has seen an increase in the proportion of brook trout between the 1970s (5.8%) and the early 2000s (30.7%; ASRD & ACA, 2009b). It is uncertain whether the increase in brook trout in the Athabasca watershed is a result of displacement of Athabasca rainbow trout resulting from competition, or replacement in situations where other factors have reduced Athabasca rainbow trout populations.

Comparatively, historical impacts of non-native species on bull trout were least severe, but its main threat in this category was non-native brook trout, which were introduced into some regions of the native range of bull trout but are concentrated in southern BC and south-western Alberta (COSEWIC, 2012). The presence of brook trout is thought to have had negative impacts on the abundance and range of bull trout throughout Alberta with greater impacts in the Saskatchewan-Nelson Rivers populations DU (COSEWIC, 2012; McLeary & Hassan, 2008; Paul & Post, 2001). In comparison, the presence of naturalised brook trout was noted in the Lower Peace River watershed in northern BC but the impact was unclear (Hagen & Decker, 2011). Bull trout and brook trout are able to hybridise but the fitness of offspring is apparently low, evidenced by low numbers of hybrids (ASRD & ACA, 2009a; Warnock, 2012). As a result, the impact on bull trout is thought to result mainly from competition. Historical declines or loss of adfluvial bull trout populations in some lakes have been associated with introductions of native lake trout (Donald & Alger, 2008). The potential historical role of brown trout on the observed bull trout range contractions and population declines is not well understood, but it is possible that they were a contributing factor (ASRD, 2012). Brown trout are not present in
most of the Western Arctic DU region (except in limited areas of the Athabasca watershed).

The ongoing and future threat posed by non-native species to each DU, as estimated by the IUCN threats calculator, is similar to historical threat levels where the highest threat level was assigned to westslope cutthroat trout, and the lowest threat level was assigned to bull trout (Figure 3). The main threat to westslope cutthroat trout in this category remains non-native rainbow trout, whereas the ongoing and future threat of brook trout has diminished from historical levels since remaining pure westslope cutthroat trout populations are constrained to high elevation sites where brook trout are currently not present and would have lower competitive ability owing to colder water temperatures (COSEWIC, 2016; De Staso III & Rahel, 1994; Paul & Post, 2001). The threat of brook trout to bull trout, however, is expected to increase under future climate change scenarios owing to the higher competitive ability of brook trout in increasingly warm waters (Rodtka & Volpe, 2007). Both non-native rainbow trout and brook trout remain a threat to Athabasca rainbow trout (COSEWIC, 2014).

Ongoing and future threats in this category include whirling disease (Myxobolus cerebralis) a salmonid parasite, which was first found in Canada in the Bow River basin, Alberta, in 2016 (CFIA, 2016). Whirling disease has had devastating impacts on trout populations elsewhere in North America (Bartholomew & Reno, 2002) with severity varying by species; of the three species, rainbow trout are the most susceptible and bull trout are the least susceptible (Hedrick, El-Matbouli, Adkison, & MacConnell, 1998). Whirling disease has since been found in other regions of Alberta but has not yet shown substantial impacts on fish populations; large numbers of juvenile trout are still readily observed (AEP, 2019b) whereas whirling disease usually results in high juvenile mortality (Walker & Nehring, 1995).

4.3 | Habitat loss and alteration

Habitat loss and alteration is a broad category of threats that can include a variety of impacts (e.g. sedimentation, loss of riparian vegetation, contaminants, eutrophication and altered flow regime etc.) and have several potential sources (e.g. agriculture, forestry, mining, oil and gas, river regulation, urban development and recreation etc.). In both westslope cutthroat trout DUs, historical logging, mining and dam development have resulted in some declines in population densities as a result of increased nutrients and contaminants (BC MOE, 2014; The Alberta Westslope Cutthroat Trout Recovery Team, 2013). In BC, coal mines are attributed with responsibility for the greatest impacts (BC MOE, 2014). In both regions, historical forest practices have resulted in removal of riparian habitat with implications for stream temperatures and sedimentation (COSEWIC, 2014, 2016). In the Saskatchewan-Nelson Rivers DUs of westslope cutthroat trout and bull trout, as well as the Alberta portion of the bull trout Western Arctic DU, most historical land-use and development occurred in the downstream reaches and likely contributed to the observed range contraction and population declines in the eastern extent of their historical ranges. While land-use practices were identified as a general threat for bull trout, historical impacts in the Saskatchewan-Nelson DU have not been quantified (COSEWIC, 2012). On the other hand, bull trout occurrence was found to be negatively correlated with percent disturbance and forestry activities (Ripley, Scrimgeour, & Boyce, 2005; Scrimgeour, Hvenegaard, & Tchir, 2008). Industrial (i.e. logging, coal mining, oil and gas exploration and extraction) and agricultural activities are thought to have had significant negative historical impacts on Athabasca rainbow trout population densities through degradation of habitat as a result of road construction, release of contaminants, loss of riparian vegetation and increased sedimentation (COSEWIC, 2014). Habitat loss and degradation have been listed among the top two threats historically impacting Athabasca rainbow trout (COSEWIC, 2014; DFO, 2018).

Habitat loss and alteration can result in habitat fragmentation, caused by dams, undersized or hanging culverts, irrigation canals or land-use practices that result in uninhabitable patches. Habitat fragmentation has different implications for species and/or life-history forms depending on their need for traversable area to meet their life-history requirements. As such, it can be argued that of the three species of interest, the migratory forms of bull trout and westslope cutthroat trout have been impacted the most by the loss of connectivity since they are each known to migrate up to 250 km between spawning and overwintering habitat (Muhlfeld et al., 2012). Indeed, bull trout in southern Alberta were reduced to 31% of their historical range before the turn of the 21st century, with disproportionate losses of the migratory form (Fitch, 1997). The migratory forms of westslope cutthroat trout are extirpated from the Saskatchewan-Nelson Rivers populations DU, but, this loss cannot be attributed to fragmentation alone (The Alberta Westslope Cutthroat Trout Recovery Team, 2013). In fact, in this case, barriers to migration may be responsible for maintaining genetic purity in the remaining upstream populations. Compared to bull trout and westslope cutthroat trout, the impact of habitat fragmentation on Athabasca rainbow trout was likely less severe owing to the small proportion of migratory fish and the lack of large dams in the Athabasca watershed. Nevertheless, hanging culverts are the primary cause of habitat fragmentation in the Athabasca watershed, which may have negatively impacted the small proportion of migratory Athabasca rainbow trout (COSEWIC, 2014). Habitat fragmentation can also have negative implications for small, isolated stream-resident populations. Metapopulation theory dictates that a regional population of a species exist as a collection of small local subpopulations that interact through dispersal (Hanski, 1998). Where dispersal is impeded by fragmentation, subpopulations can become less viable through inbreeding and/or may be more susceptible to stochastic events (Hanski, 1998). Metapopulation dynamics may be important for stream-resident populations in particular for small patch sizes (Hilderbrand & Kershner, 2000; Kovach, Armstrong, Schmetterling, Al-Chokhachy, & Muhlfeld, 2018; Riemann & Dunham, 2000), but persistence of some isolated westslope cutthroat trout populations suggests that risks to viability when faced with isolation may not be


as large as once thought for this species (Peterson, Rieman, Horan, & Young, 2014; Rieman & Dunham, 2000).

Overall, the westslope cutthroat trout Saskatchewan-Nelson Rivers populations DU has the lowest ongoing and future threat levels related to habitat loss and alteration since they currently occupy such a small area and are limited to high elevation headwater stream reaches where further development is either unlikely or not permitted (Figure 3). At the other extreme, the Athabasca rainbow trout DU faces the highest overall threat levels related to habitat likely reflecting their position in a watershed with substantial ongoing industrial activity. Threats from specific sources (e.g., energy production and mining, pollution, natural systems modification which includes river regulation) were highest for Athabasca rainbow trout and the Pacific populations of westslope cutthroat trout, reflecting the higher occurrence of oil and gas and/or mining operations, other industrial activities and the likelihood that development will continue in those regions (BC MOE, 2014; COSEWIC, 2014). On the other hand, most of the potential future development in the Saskatchewan-Nelson Rivers populations DUs is likely to occur in areas where westslope cutthroat trout and bull trout have already been extirpated. The impact of transportation corridors (specifically development of new roads) and human disturbance (mainly recreational activities e.g. off-highway vehicle (OHV) use) was thought to be highest for Athabasca rainbow trout and bull trout (Saskatchewan-Nelson Rivers populations). The difference between habitat threat scores for bull trout and westslope cutthroat trout in south-western Alberta reflects the already constrained distribution of westslope cutthroat trout.

4.4 | Climate Change

Assessing past impacts of climate change on species distributions and population densities are hindered by relatively short timescales for which fish data exist and confounded by co-occurrence of most land-use alterations in areas where climate change effects would be most evident. For example, land-use alterations and climate change effects were both more prevalent in the prairie region of southern Alberta as opposed to the mountainous headwater region. Based on species’ biology, it is likely that past climate change has impacted bull trout the most, followed by westslope cutthroat trout, and lastly Athabasca rainbow trout. It is probable that at least some of the range contraction observed in the Saskatchewan-Nelson Rivers DUs of westslope cutthroat trout and bull trout are due to past climate change. That westslope cutthroat trout still occupy the Pacific populations historical range suggests that the historical impact of climate change was less severe relative to that experienced in south-west Alberta. Climate change was not noted as a threat to the Western Arctic bull trout DU likely because of its northern position and that climate change impacts on bull trout are expected to be greatest at the southern extent of the species’ range (Dunham, Rieman, & Chandler, 2003). Still, it should be noted that past climate change has probably played a role in the observed range contraction and population declines in the Alberta section of the Western Arctic DU (Isaak et al., 2012). Climate change is not thought to be an immediate concern for Athabasca rainbow trout likely owing to their higher water temperature thresholds (COSEWIC, 2014).

Based on their cold water requirements (Table 2) and that they still occupy the foothills areas of Alberta, bull trout face the largest imminent threat from climate change. Bull trout are also more susceptible to predicted increases in winter flows, which has been linked to reduced recruitment for the autumn spawner owing to redd scouring (Isaak et al., 2012). In comparison, westslope cutthroat trout in south-western Alberta face a lower threat from future climate change since they are constrained to upper elevation waters, which may provide refuge from increasing water temperatures for some time (COSEWIC, 2016). Although Athabasca rainbow trout are the least susceptible to increasing water temperatures, they are likely susceptible to predicted changes to the magnitude and timing of flow. Athabasca rainbow trout spawn immediately following the spring freshet and position their redds in a stream channel depending on water depth and velocity (Sterling, 1992). In years where spring flow is higher than average, females may build redds closer to the channel margin, making eggs more susceptible to desiccation during low summer flows. On the other hand, in years of low spring flows, females may build redds closer to the centre of the channel, making the eggs more susceptible to scour (Sterling, 1992).

<table>
<thead>
<tr>
<th>Species</th>
<th>Designatable unit</th>
<th>Calculated overall threat level</th>
<th>Assigned overall threat level</th>
<th>Generation time (years)</th>
<th>Three-generation population decline projection (%)</th>
<th>Median</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Westslope cutthroat trout</td>
<td>Pacific</td>
<td>High-very high</td>
<td>High</td>
<td>5</td>
<td>40</td>
<td>10–70</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Saskatchewan-Nelson Rivers</td>
<td>High-very high</td>
<td>High</td>
<td>5</td>
<td>40</td>
<td>10–70</td>
<td></td>
</tr>
<tr>
<td>Bull trout</td>
<td>Saskatchewan-Nelson Rivers</td>
<td>High-very high</td>
<td>High-very high</td>
<td>10</td>
<td>55</td>
<td>10–100</td>
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<tr>
<td></td>
<td>Western Arctic</td>
<td>Not completed</td>
<td>Not completed</td>
<td>–</td>
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<td>–</td>
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</tr>
<tr>
<td>Rainbow trout</td>
<td>Athabasca River</td>
<td>Very high</td>
<td>Very high</td>
<td>5</td>
<td>75</td>
<td>50–100</td>
<td></td>
</tr>
</tbody>
</table>
5.1 | Temporal patterns in threat severity

The severity of individual threats has changed over time. Whereas angling once represented the largest threat for some DUs, it is now arguably the lowest threat throughout the region. This change can be attributed to implementation of more restrictive angling regulations and is responsible for some remarkable recoveries (e.g. Johnston et al., 2007; Parker et al., 2007). Despite substantial improvements, it has been suggested that the threat remaining from angling (i.e. from poaching, accidental misidentification and post-release hooking mortality) may still result in unsustainable populations (Post et al., 2003), particularly in combination with other threats. For example, post-release hooking mortality is known to increase with increased water temperature (Bartholomew & Bohnsack, 2005), and the disproportionate catchability of bull trout and cutthroat trout relative to brook trout and non-native rainbow trout may result in additional post-release hooking mortality for native trout (MacPhee, 1966). In fact, a study that looked at the impacts of a selective harvest programme in Quirk Creek, south-western Alberta, illustrated low impact of selective harvest on non-native brook trout while exposing native trout (bull trout, westslope cutthroat trout) to further risk (Paul, Post, & Stelfox, 2003). So, although recent restrictive regulations were designed to sustain populations of popular recreational fishes, it is possible that the sum of post-release hooking mortality and poaching may render these residual populations unsustainable in the face of multiple cumulative threats.

Undoubtedly, climate change effects have already impacted these native cold-water fish; however, this impact is expected to intensify in the future with a 3–5°C increase in water temperatures predicted between 1999 and 2099 (Christensen et al., 2007; Isaak et al., 2012). Mean July water temperatures in the South Saskatchewan River basin in south-western Alberta, for example, are already between 10 and 12°C in the headwaters and up to 14°C in the foothill region where bull trout are still found (Mee, Robins, et al., 2012). Mean July water temperatures in the South Saskatchewan River basin in south-western Alberta, for example, are already between 10 and 12°C in the headwaters and up to 14°C in the foothill region where bull trout are still found (Mee, Robins, et al., 2012). Projected temperature increases will further restrict the bull trout range, impact interactions with non-native trout that are better suited to and more competitive in warmer waters, and increase the risk of post-release hooking mortality (Bartholomew & Bohnsack, 2005; Paul & Post, 2001; Rodtka & Volpe, 2007; Warnock & Rasmussen, 2013).

Comparing the magnitude of the threat posed by non-native species between historical levels and future projections is complex. On the one hand, only sterile non-native species are now stocked in these regions, so the threat of future government-mandated introductions of non-native species has been reduced. Nevertheless, naturalised populations of non-native species are an ongoing problem. In many cases, the range of naturalised non-native species has expanded beyond the sites where they were originally stocked and in some cases, non-native species have completely replaced or displaced native species (COSEWIC, 2014; Paul & Post, 2001). Non-native species can out-compete native species in some situations. For example, it has been suggested that brook...
trout are better suited to the small groundwater-fed streams inhabited by Athabasca rainbow trout (ASRD & ACA, 2009b). Since brook trout are autumn spawners, they are not vulnerable to substantially varying spring flow levels that leave Athabasca rainbow trout eggs susceptible to desiccation or scouring (Sterling, 1992). Further, brook trout are known to build redds on groundwater upwelling sites to prevent freezing (Blanchfield & Ridgway, 1996; Curry & Noakes, 1995) and brook trout are capable of digging through fine sediment to build redds (Pépino, Franssen, Rodríguez, & Magnan, 2012), making them less vulnerable to habitat degradation from anthropogenic activities or stochastic natural events (ASRD & ACA, 2009b). New invasive species or those with yet unknown impacts (e.g. whirling disease, zebra mussels) also make future impacts difficult to predict.

Similar to the threat of non-native species, assessing the change in severity between historical and future threats from habitat loss and alteration is complex. In south-western Alberta, most future habitat alteration is unlikely to occur in high mountain elevations and park areas where bull trout and westslope cutthroat trout remain; thus, the future threat from habitat loss and alteration is considered low. The one exception to this is habitat destruction that can arise from some recreational activities (e.g. OHV use, horseback riding, etc.). In comparison, ongoing development within the other DUs means that the threat of habitat loss and alteration may stay the same or worsen. It is important to note that some activities have improved practices over time, resulting in reduced degradation. For example, riparian buffers are now left in forestry best practices limiting negative habitat impacts on streams (Warrington et al., 2017). As well, work has been done to identify the impacts of grazing on riparian and stream environments, and encourage ranchers to improve cattle grazing practices by using streambank fencing and off-stream watering systems (Ambrose, Fitch, & Bateman, 2006; Fitch & Adams, 1998; Miller, Chanasyk, Curtis, Entz, & Willms, 2011). As development occurs in new areas, however, access for angling and recreational activities increases, which is an indirect impact of development despite improvements in industry best practices (Gunn & Sein, 2000a).

**5.2 | Mechanisms driving observed differences: geography versus biology**

The relative magnitude of each threat category (both historical and future threats) varied among DUs and it can be argued that these differences reflect differences in the location of the DU, species-specific biology or a combination of the two. Which factor (i.e. geography vs. biology) plays a larger role depends on the threat category. For the most part, the magnitude of the threat posed by habitat loss and alteration depended on the region inhabited by the DU and the amount and type of anthropogenic activity occurring rather than a specific susceptibility of a species dictated by its biology. The one exception to this is habitat alteration that results in fragmentation because of disproportionate impacts on the migratory life-history forms. On the other hand, the severity of the threat posed by non-native species is related primarily to species-specific characteristics. The magnitude of the threat posed by non-native species is different for each main native/non-native pair (i.e. bull trout/brook trout, westslope cutthroat trout/non-native rainbow trout, Athabasca rainbow trout/non-native rainbow trout) and varies along a gradient depending on the mechanism of the biological interaction. Among our examples, the threat was lowest when the interaction between species was primarily characterised by competition (e.g. bull trout vs. brook trout). Although hybridisation between bull trout and brook trout is known to occur, it does not appear to play a predominant role; rather competition is the main interaction between these species and is mainly mediated by environmental variables (McMahon, Zale, Barrows, Selong, & Danehy, 2007; Rodtka & Volpe, 2007; Warnock & Rasmussen, 2013). Brook trout have a demographic advantage owing to their much earlier age-at-maturation and rapid life-history (i.e. fast growth, early reproduction, short life span; Paul et al., 2003), however, in areas where bull trout retain their competitive ability due to cooler temperatures, bull trout are able to remain dominant (Warnock & Rasmussen, 2013). On the other hand, the threat from non-native species was greater where hybridisation played a substantial role. Indeed, hybridisation has been known to increase displacement rates by non-native species (Huxel, 1999). In our examples, the threat was greatest when hybridisation was coupled with competition between two distinct species (e.g. westslope cutthroat trout vs. non-native rainbow trout). The threat was intermediate when hybridisation was coupled with intraspecific competition (i.e. Athabasca rainbow trout vs. non-native rainbow trout).

The threat posed by angling depends heavily on the biology/ecology of the species. For example, bull trout are particularly susceptible to angling because of their high catchability (Prince, 1912) and are likely more susceptible to post-release hooking mortality because of their temperature sensitivity (Boyd, Guy, Horton, & Leathe, 2010). However, geographic location plays a role as well since accessibility of streams and proximity to population centres are important drivers of fishing effort (Carruthers et al., 2019; Gunn & Sein, 2000b). In addition, the late age-at-maturation and large body size render the adfluvial and fluvial bull trout particularly susceptible to overfishing as they are vulnerable to fishing even as immature fish (Post et al., 2003).

The threat of climate change depends on both geography, where access to high elevation or groundwater zones may provide refuge from increasing temperatures, and biology, where some species are more susceptible to increases in temperature and changes to precipitation predicted by climate change. Climate change poses the greatest direct threat to bull trout owing to the species’ low and narrow thermal niche (Wenger et al., 2011). However, the threat is likely less severe for much of the Western Arctic DU owing to its low temperatures at present. Bull trout may find refuge in cold, high elevation reaches within its current range even in south-western Alberta. Conversely, Athabasca rainbow trout would not be able to benefit from thermal refugia in the high elevation colder waters of Jasper National Park owing to non-native rainbow trout dominating that area (COSEWIC, 2014).
5.3 | Managing variable threat profiles

The future threat severity profile (i.e. the number of low, medium, high, and very high threats for each DU) varies among DUs (Figure 3). The two extreme examples are the westslope cutthroat trout Saskatchewan-Nelson Rivers populations DU, which was characterised by mainly low threats and one high-very high threat (non-native species), and the bull trout Saskatchewan-Nelson Rivers populations DU, which was characterised by multiple mid-level threats and its highest threat ranked at medium-high (climate change). Although resulting in similar calculated overall threat impact among DUs (Table 3), the different distribution in severity among threat categories has implications for management. The first scenario (westslope cutthroat trout Saskatchewan-Nelson Rivers populations DU) where the threat facing a species comes primarily from a single identifiable source, in theory, presents a more manageable problem; all efforts can be targeted to resolve just one problem. In the situation facing bull trout where threats are individually less acute but are many, prioritising management efforts among threats is more of a challenge (Craig et al., 2017).

Not all threats are equally manageable, which also has important implications for recovery management. Historical angling is a good example of a threat that was easily managed through imposition of restrictive regulation. On the other hand, climate change is a global issue that cannot be addressed by making changes to jurisdictional policy alone. Managing threats is further complicated when two threats have opposing solutions. An example of this is simultaneously managing the threats of fragmentation and non-native species. On the one hand, fragmentation is responsible for low or extirpated migratory populations of westslope cutthroat trout and bull trout and isolating some small, stream-resident populations with implications for metapopulation dynamics, but it is also responsible for maintaining pure populations of stream-resident westslope cutthroat trout. In these situations, trade-offs must be assessed and policy choices among populations must be made (Fausch, Rieman, Dunham, Young, & Peterson, 2009).

5.4 | Inherent challenges to threat comparisons

A qualitative comparison of threats across species and regions, such as was conducted here, provides a means to compare population statuses and severity of the threats facing each species/DU. However, it is complicated by inherent challenges that result in an imperfect assessment that is not void of subjectivity. While a strength of the species at risk assessment process in Canada is the use of a single national body to conduct COSEWIC assessments (Waples, Nammack, Cochrane, & Hutchings, 2013), the COSEWIC status reports do not provide the context needed for interspecific comparisons and is even limited in supporting comparisons among DUs. The comparisons we conducted here were made more challenging by compiling data and/or expert opinion from sources that used a variety of methods or different reference standards. Reference values for westslope cutthroat trout in the Pacific populations DU as well as all three species in Alberta were derived from some of the most productive systems in each region, resulting in watersheds being classified as low density without consideration for natural limits of each watershed and human activity implied as the cause. The reference densities used for BC bull trout based on US criteria are likely to be even more unrealistic. While likely the most conservative option when historical site-specific data are unavailable, the use of questionable reference population densities may present an inaccurate picture of the current state or result in wasted resources if attempting to restore populations to unfeasible levels. Lastly, the quantity of data available to determine population statuses varied among regions raising issues of comparability and the confidence with which low data regions can be viewed.

The difference in how genetic integrity is currently considered among the DUs also creates bias. The westslope cutthroat trout Saskatchewan-Nelson Rivers populations DU is the only DU that excludes populations containing hybrids in the total count of remaining populations. As a result, a cursory comparison of the assessment reports suggests a much worse situation for the Alberta westslope cutthroat trout relative to all other DUs. This difference, however, is exacerbated by the way genetic information is included and highlights the importance of consistency in addressing introgressive hybridisation when determining conservation status (e.g. Allendorf et al., 2004).

Understanding the overall threats situation is also hindered by challenges in identifying the right threat when there are multiple possibilities. For example, in the historical threat assessment for both Saskatchewan-Nelson Rivers populations DUs, all four threats categories (i.e. angling, non-native species, habitat loss and alteration, and climate change) likely contributed to the declines at the downstream reaches of the range. Determining the relative roles these threats played is difficult, if not impossible. This issue is further complicated when threats interact to produce outcomes that are synergistic or antagonistic so that the magnitude of the impact is unexpected (Craig et al., 2017). An example of this is where increased temperatures associated with climate change will have direct impacts on the range of bull trout because of their biological constraints but will also have indirect effects through enhancing the competitive ability and distribution of brook trout.

6 | CONCLUSION

An exploration of historical and future threats facing native trout and charr in the Canadian Rocky Mountain region presents a bleak portrayal of the current status and future persistence of these fish species in the region, and indeed calls for immediate action. Future threat levels point to a high probability of substantial continued declines without intervention and for some DUs even understates the risk. Efforts to reinforce current populations or recover extirpated populations will require development of conservation
priorities, identification of feasible recovery and conservation strategies and likely the use of management techniques that are novel to the region. Several techniques have been employed in other regions to tackle similar issues and provide new opportunities and risks that should be evaluated. Intentional installation of barriers has been used as a management strategy to prevent invasion by non-native species (Rahel, 2013) but comes with costs of isolation and fragmentation that may have detrimental long-term side effects for the target populations and other native populations (Fausch et al., 2009; Lusardi, Stephens, Moyle, McGuire, & Hull, 2015; Peterson, Rieman, Dunham, Fausch, & Young, 2008). Implementation of freshwater protected areas has been suggested as a means to protect freshwater environments from a number of threats at once, providing a space where disturbances are minimised (Crivelli, 2002). Freshwater protected areas could provide an immediate refuge from angling and habitat destruction (Suski & Cooke, 2007) but are not, on their own, effective in addressing non-native species impacts or climate change effects (Chessman, 2013); they may, however, be coupled with non-native species eradication methods and conservation translocations to promote native trout recovery (Clancey et al., 2019). Conservation translocations, the deliberate movement of living organisms from one area to another for conservation purposes, are being applied increasingly in an effort to restore native fish populations with mixed results (Galloway, Muhlfeld, Guy, Downs, & Fredenberg, 2016; Hayes & Banish, 2017; IUCN, 2013). The viability and risks associated with translocations are complex and several factors should be considered including the genetics of the host population and potential transference of disease (IUCN, 2013). Success of these programmes can be unpredictable, but generally the threats that lead to the initial declines should be addressed prior to translocation to maximise likelihood of success (Hayes & Banish, 2017).

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