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Natural Resource Stewardship and Science

Glacier National Park Stream Ecological Integrity

2007–2019 Trend Synthesis Report

Natural Resource Report NPS/ROMN/NRR—2024/2631





ON THIS PAGE
Rocky Mountain Network stream crews access a North Fork of the Flathead River sentinel site, Glacier National Park, 2008.
NPS / BILLY SCHWEIGER

ON THE COVER
Glacier National Park staff collect long term monitoring stream data from Reynolds Creek, Glacier National Park, 2018.
NPS / J. ASEBROOK

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Executive Summary

Background and introduction

Glacier National Park has some of the cleanest water in the world. The park's streams and rivers are treasured for their beauty and pristine condition. In addition to offering awe-inspiring views, these ecosystems are home to a globally significant community of animals, including imperiled bull trout (*Salvelinus confluentus*), west-slope cutthroat trout (*Oncorhynchus clarkii lewisi*), several endemic macroinvertebrates (e.g., the threatened meltwater stonefly (*Lednia tumana*)), and almost 50 new to science diatom (algae) species added to the books since 2007. Streams and rivers are often what visitors come to see and what they remember when they leave. They have played an important role in the history and culture of the region.

Streams are one of the most threatened natural resource types worldwide—even in large wilderness parks like Glacier. Biologists estimate that 95% of Montana's waters are degraded, have lost native species, and/or have been invaded by nonnative species. Though rare in the park, human stressors are often factors driving these impacts. Anthropogenic impacts can be both direct and indirect, including impoundments, diversions, channelizing, rip-rap, culverts, de-watering and pollution, introduction of aquatic invasive species, and wildland fires that affect riparian cover and sediment input. Cumulative impacts from larger scale disturbances such as forestry, mining, air pollution, and climate change are also important added threats that do not recognize boundaries of protected areas like Glacier.

Streams and rivers are excellent indicators of Glacier's overall ecological condition. The National Park Service Rocky Mountain Inventory and Monitoring Network (hereafter, network) has selected a suite of stream indicators that provide long-term ecological monitoring data and information to park managers. Traditional stream monitoring approaches focus on large changes in water chemistry or hydrologic regime, which in general do not occur in the park. Rather, network monitoring is based on the premise that understanding and managing the streams and rivers in the park is best accomplished by treating them as complete hydrologic systems. These hydrologic systems, when in good condition, support a balanced, integrated, and adaptive community of organisms, with a species composition, diversity, and functional organization comparable to that of the highest quality natural habitats of the region. To accomplish this, we use an integrated approach called bioassessment. While regulatory considerations or site-specific project considerations provide an important context for stream monitoring in Glacier, our emphasis on long-term ecological integrity is an important distinction from stream monitoring conducted by the state, other federal agencies, or tribal nations.

This report summarizes Stream Ecological Integrity (SEI) monitoring at six sentinel stream and river sites in Glacier from 2007 to 2019. An earlier report provided park-wide baseline results from a large statistical sample of streams in 2007–2009. We focus in this document on estimating and assessing trends in chemistry, habitat, and biology at Glacier's sentinel stream sites.

Navigating the report for different audiences

This report provides both high-level summaries and comprehensive detail to meet the needs of its two primary audiences: park resource managers and network and collaborator scientists. Managers needing a concise overview of results can read this Executive Summary, as well as summary

condition statements (in tables) throughout the report. Details are in the narrative and appendices to provide in-depth context and methodological guidance.

Methods

SEI methods are documented in the *Rocky Mountain Network Stream Ecological Integrity Monitoring Protocol.* They are derived from well-established and existing methods, many of which were developed by network partners. The application of a standardized monitoring protocol facilitates comparisons of the park's streams and rivers to those within the same ecoregions as the park.

From 2007 to 2019 we sampled four stream sentinel sites, three or four times annually. Two additional sites were added at request of Glacier in 2018 to augment information for important large rivers designated as Wild and Scenic. Sentinel sites were selected to provide an indication of how stream conditions change over time at specific stream locations. Over the period of record included in this report, we conduced 63 full events at these sites where all methods were implemented, and an additional 129 partial events where we collected chemistry samples. *Sample design summary* in the Methods section of this report provides more sample design details and maps of sample sites.

Components of the stream ecological integrity protocol

The SEI protocol includes a broad spectrum of indicators to help evaluate the ecological integrity of the streams and rivers of Glacier. We include measures of both drivers and stressors and ecological responses such that we better understand the linkages between these and help parks apply these relationships to resource management. Core indicators include multiple measures of water and sediment chemistry (i.e., nutrients, metals, temperature), physical habitat (i.e., substrate composition, riparian cover, channel morphology), and assays of two important biological assemblages: macroinvertebrates and algae.

Human disturbance

One of the foundations of SEI monitoring is the estimation of human disturbance(s) that potentially impact Glacier's stream condition and degrade ecological integrity. We use a Human Disturbance Index (HDI) to estimate reach-scale human stressors and catchment-scale disturbances at each monitoring site. Key stressors in the HDI include roads and trails, structures, and land use, such as forestry and mining (adjacent to the park). The HDI allows incorporation of historical impacts that may still matter to a stream's condition.

Water chemistry

Water chemistry (often described with the more general term "water quality") is a key element of stream habitat. It is also a common focus of most long-term stream monitoring programs, especially for regulatory purposes under the Clean Water Act. Water chemistry includes parameters analyzed in the field, including dissolved oxygen, pH, specific conductance, and water temperature sampled with probes. We also include parameters measured in the lab: nutrients, metals, sediments, and major ions. Stream water chemistry, especially nutrient levels, can be affected by human disturbances.

Physical habitat

Physical habitat is an important control on ecological processes and ecological integrity in Glacier's streams. Physical habitat of streams includes stream channel dimensions and patterns (e.g., width and depth, ratio of riffles to pools, gradient), sediment characteristics (the size and types of sediments that make up the streambed), amount and type of wood in the stream, stream flow, and the amount and type of vegetation in the riparian zone. These characteristics influence stream habitat volume, complexity, interaction with the floodplain, and other important determinants of the species and distribution of plants and animals that live in and near the stream. Stream physical habitat, especially sediment levels and stream morphology, like pool depth, can be affected by human disturbances.

We include invasive/exotic plants as an important component of our physical habitat monitoring. Understanding and monitoring invasive/exotic taxa in riparian systems is key for Glacier resource management. Invasive/exotic riparian taxa can be associated with multiple negative effects, including 1) alteration of fire, nutrient, or hydrologic regimes; 2) changes to patterns in carbon sequestration; and 3) changes to plant and animal distributions.

Bioassessment

Macroinvertebrates and algae are major components of the biodiversity found in streams. They are the base of the food web and thus support a broad range of other species. They are useful and efficient indicators of ecological integrity because of their ability to integrate different processes and because they respond to physical, chemical, and biological disturbances at multiple temporal and spatial scales. The diversity of Glacier's macroinvertebrate and algae communities is exceptionally high, with a continentally outstanding biological distinctiveness. In this report we assess macroinvertebrate and diatom communities in relation to specific aspects of human disturbance, such as nutrient concentrations, fine sediments, or water temperature. In a general sense, these indicators tell us if a particular stressor's values outside a reference range "matter" to a sentinel stream in the park.

Aquatic invasive species

Aquatic invasive species can have a strong effect on stream ecological integrity and are a focus of park resource management. We surveyed for zebra mussel and quagga mussel and New Zealand mud snails. We also recorded presence and extents of blooms of the invasive but native diatom, rock snot (also known as didymo). Rock snot can form extensive mats, inhibiting the growth of other algal species, changing the composition of aquatic communities, decreasing suitable spawning habitat for fish, changing in-stream chemistry, and lowering the aesthetic appeal of streams.

Assessment

A key element of SEI monitoring is the interpretation of results in a way that is both scientifically defensible and relevant to the management of Glacier's streams. In broad stroke, we compare our monitoring results to numeric "assessment points" that define disturbance or condition classes. Assessment points can be set based on management guidance, regulatory criteria, meaningful ecological thresholds, and values derived from reference sites in the park and/or its surrounding ecoregions. We interpret comparisons between SEI data and these assessment points in a collaborative way with park staff and management in the context of park management objectives,

National Park Service Inventory and Monitoring Division (IMD) guidance, National Park Service resource management policies, and general ecological theory.

Results

Overall, the ecological integrity of sentinel stream sites in Glacier remained in a reference (good, with only minor disturbance) condition from 2007 to 2019. Our data do not suggest that the largely pristine water quality or biodiversity of these streams has changed in any profound way. Nevertheless, emergent issues may be beginning to reveal themselves in our data. These are presented below.

Emergent issues

Climate change

Climate change is one of the most pressing issues impacting streams in relatively undisturbed parks like Glacier. The park's remoteness, its large size, and a generally intact current stream condition will probably not insulate important stream resources from broad scale changes in climate moving into the future. Specifically, climate change will make it more difficult to achieve water chemistry standards in the park, especially for water temperature, hydrology, and sediments (and/or responses linked to these). Furthermore, many aspects of the biological communities in streams will be directly and indirectly impacted by changes in temperature, precipitation, and the overall water cycle.

Our data suggest that climate driven processes may be negatively affecting select aspects of sentinel streams in the park. It's likely that this has been happening for some time, as revealed by other research, and our data are just now becoming long enough to see changes. This is most clear in our water temperature data, where we see an increasing trend of warming temperatures at several sites from 2007 to 2019. Our models develop connections between the water cycle in watersheds and increasing temperatures, suggesting that multiple complex factors may interact to create warmer water—growing degree days, evapotranspiration, reduced water available for the water cycle ("climatic deficit"), and other factors. There is likely a cascade of impacts from warmer waters in streams as summarized throughout this report. Climate is the master variable controlling, at least in part, many aspects of the stream ecology in Glacier. However, and importantly, climate driven changes are often subtle, almost always complex, and at times unpredictable. It is important to continue monitoring to better understand cause and effect relationships.

Water chemistry

While most chemistry responses at sentinel sites are in a reference condition, elevated water temperatures and altered hydrologic processes may be impacting stream chemistry. For example, metals in stream sediments were at unexpected levels in our alpine sentinel site, Reynolds Creek, perhaps driven by complex climate change processes. There are also some indications that atmospheric deposition of nitrates is interacting with climate driven changes, impacting pH and alkalinity, and that visitor services may have some possible effects at specific sites by elevating some major ions. *We need additional data, analysis, and interpretation to confirm and support these patterns. We recommend the park partner with the Inventory and Monitoring Division and/or external collaborators to conduct focused research on a possible climate change driven acidity and metals issue at the Reynolds Creek site, as what may be happening in this stream could be happening*

at a landscape scale in the park and therefore merits additional investigation. In addition, more focused research on possible visitor service driven impacts on water quality at Swiftcurrent Creek may be warranted.

Habitat

Most of our habitat indicators suggest sentinel stream site habitat is intact and in a reference condition. There may be some fine sediment issues at some locations and, using a conservative assessment point, the frequency of invasive/exotic plants was in a non-reference condition at all sentinel sites. Perhaps the most important missing piece in the habitat elements of the SEI protocol is a lack of management relevant and park specific assessment points. *We recommend the park partner with IMD and external collaborators to conduct focused research on deriving or better understanding existing assessment points for habitat features like invasive plants, fine sediment, and wood density. In general, any improvement in the connection between park resource management and the assessment points used to evaluate data would enhance the relevance of long-term monitoring.*

Bioassessment

Overall, our bioindicators showed that sentinel sites were in largely reference conditions. In some cases, a site's biology allowed us to reconsider and adjust assessments based on chemistry or habitat data alone. We generally consider biological responses as more important—if a stream had intact biology, it likely suggests that most physical or chemical stressors may not really be impacting the site. Nevertheless, bioassessment also revealed results of concern to the park, specifically a negative biological response to warming water, with ecological integrity higher in colder water. *We recommend the park partner with IMD and external collaborators to conduct focused research on temperature mediated impacts to biological communities in the park. Specifically, while we did not see the feared warm water driven "didymo take over" of sentinel sites in the park, we do not have the complete story on this native but invasive diatom. A focused research effort on didymo would give us a better understanding of its potential impacts to the park.*

Management applications

A fundamental goal of Rocky Mountain Network inventory and long-term monitoring is to provide park managers with data and information useful for protecting and managing park resources. Our estimates of trend in stream condition at sentinel sites can help in the understanding of how and what changes have occurred over the last decade and how this might translate to future conditions. This information can help park management report on and meet regulatory requirements and assist with management in the park. Our data can be used to quantify the importance of various stressors or drivers of stream condition and their trends over time (e.g., nutrient input, sediment loads, climate change-driven changes in stream hydrology, etc.). Specific indicators, when interpreted in the context of the larger SEI dataset, can also enhance general understanding of stream ecology. Finally, SEI results should also help park staff understand and communicate to the public about the condition and trend over time of stream resources in Glacier. They allow interpretation of specific streams in the context of the broader resource.

Perhaps most importantly, network stream monitoring at sentinel stream sites provides a critical ecological baseline and estimate of the trajectory of change (trend), both of which are invaluable for future assessments at these sites and, with careful modeling, across the park. As just one example, the park's namesake glaciers—such an important part of stream ecology—are projected to be gone by around 2030 (just seven years from this report's publication!). This ongoing change is accelerating increases in stream temperatures, shifts in hydrologic regimes, and sediment loads. SEI monitoring is documenting and will continue to document reduced stream flow, channels with less habitat, altered sediments, and warmer water temperatures. All these changes will likely have detrimental impacts to aquatic communities. We intend that network data will offer the park useful guidance on how to interpret and manage these systems in the face of these changes.

Future stream monitoring

This effort has been, and will continue to be, a cooperative undertaking between the network and our partners—most importantly, Glacier National Park. We believe our results, representing the efforts of dedicated scientists and resource managers over several years, address our objectives, are valid, representative, and in many cases, actionable. However, our data are still relatively "short," and we are still only at the beginning of this effort, even a decade or more in. It will take many more years to have sufficient data to fully understand long-term trends. As we accrue data, we will also continue to improve our understanding of the mechanisms and drivers behind SEI responses. These analyses will help scientists continue to monitor the most cost-effective and management-relevant suite of indicators of ecological integrity of the streams and rivers of Glacier National Park.

Monitoring complex ecological systems like streams and rivers is always a challenge. This is even more so in landscapes like Glacier that are generally intact, with few classic human disturbances that lead to clear and unambiguously degraded streams. Assessment methodology in many ways depends on these structural relationships between stress and response to classify the state of streams in the park and their direction of change. However, a clear role for the park and our monitoring data might be to serve as a reference for other streams embedded in more anthropogenic landscapes. Our results can help managers of streams and rivers not lucky enough to be in Glacier aspire to maintain or be restored to a similar condition.

Finally, recent work suggests impact from climate change on biological condition may be offset if other stressors on a system are reduced. This is, in a sense, paying climatic debt with accrued environmental credit. Systems like Glacier National Park are of course well positioned to do this—the park already has a rich bank account full of environmental credit given its wilderness context and its largely reference condition. This offers a path forward for park resource management and recognizes that the park's waters remain largely in a pristine state.

Summary condition

Summary condition statements are presented throughout this report as syntheses of our most important results. The network's approach to summary condition includes enough detail that a resource management audience might focus only on the summary condition statements in this report to glean both the most important results and some key context and description.

A summary condition statement includes supporting narrative and a symbol designed to convey with a glance the condition and trend in an indicator and our confidence in the assessment. We present high-level summary condition results for classes or groups of indicators monitored at GLAC from 2007 to 2019 in Table ES1. Finally, summary condition statements with more detailed and explanatory text begin each chapter in the main body of the report.

Summary condition statements should be used with some caution. Many indicators have complex relationships with assessment points. We also often combine indicators in a category (to simplify high level reporting). Thus, we are sometimes generalizing across multiple indicators, where one may be in a reference state, and another may not. This is also true across sites, and we often must make general statements that synthesize patterns that might diverge across sentinel streams. As possible, we highlight site specific patterns in the narrative, but for simplicity we interpret overall patterns across sites in most summary condition statements.

Table ES1. High level summary condition statements by major stream vital sign category, Glacier National Park sentinel sites, 2007 to 2019. Hyperlinks connect to more detailed summary condition statements in the body of this report, which themselves synthesize the full analytical results. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value (i.e., increasing temperatures are usually signs of decreasing condition and the arrow thus would point downwards).

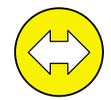
Vital sign	Summary condition	Status, trend, and confidence
Climate and water balance	Climate and water balance play an important role in the ecology of Glacier's streams and rivers, and understanding climatic patterns can help interpret stream condition. Based on the historical and future climate expectations, Glacier should expect warmer and slightly drier or more variable summers and warmer, wetter winters. Coupled with current and expected future warming, water cycle models suggest available water for resources will likely decrease over time. There is some variability in trends across east to west and elevational gradients, with complex ramifications on streams and other park resources. Given overwhelming evidence in the literature and in our general models, we score climate and water balance as in a non-reference condition, with a decreasing trend.	
Human disturbance	Stream ecological integrity monitoring includes an estimate of human disturbance(s) in and around a stream that potentially impact condition. We use a Human Disturbance Index (HDI) that combines multiple classes of disturbances at a site with watershed scale stream–road crossing density. No sentinel site HDI value was outside the assessment point over our period of record, suggesting sites were in a reference condition relative to human disturbance. However, five of the six Rocky Mountain Network sentinel sites are in the front country or immediately adjacent to the front country and have at least some measurable human disturbances. There were no significant trends in HDI at any sentinel site over our period of record. Field data indicate potential disturbance from roads increased at the North Fork of the Flathead border and Swiftcurrent Creek over our period of record. Given the high level of visitation in the park and some indication of increasing levels at stream sentinel sites, human disturbance warrants some caution at select sites.	
Water and sediment chemistry	The chemical environment of stream water and sediment is the primary medium aquatic biota inhabit. Documenting the status and trends in chemistry is a fundamental part of monitoring streams and this has been done for decades within multiple long-term ecological monitoring programs. Most trends in water and sediment chemistry at Glacier sentinel sites were largely flat, with most parameters changing little over the roughly decade we analyze in this report. Moreover, most parameters were within a reference range, meaning concentrations were not beyond assessment points established to classify status and trend at sentinel stream sites in the park. These results are as expected for the Outstanding Resource Waters (ORW) of Glacier National Park. However, there are a few exceptions to this, with some potentially emergent issues of importance. Metals are at unexpected levels at our alpine sentinel site, likely driven by complex climate change processes. There are also some indications that deposition of nitrates is interacting with climate driven changes, and there are some limited possible effects at specific sites from visitor services that may be elevating some major ions. All of these subjects need additional data, analysis, and interpretation; however, it is likely that they are all connected to elevated water temperatures.	
Water temperature	Water temperature is a critical driver of stream ecological condition, affecting the physiology of plants and animals. Thus, it is a key parameter monitored in Glacier National Park. There were significant increasing trends in water temperature at most of our sentinel sites. Warmer water generally implies deteriorating conditions, as cold water organisms are intolerant of warmer water. Our models suggest increases in water temperature may be accelerating; currently they are on the order of 1 to 1.5 °C a decade. Many summer stream temperature daily maxima were above ecological assessment points based on aquatic invertebrates or bull trout habitat suitability. Bioassessment supports these general conclusions. Temperature-based macroinvertebrate bioindicators showed the effects of warming water at most sites. Water temperatures have the most direct and clearest connection to climate change. In Glacier, where climate change is a pressing issue, these dynamics are even more evident. Water in Glacier, as the name suggests, is influenced by the park's iconic glaciers. As melt-induced flows are reduced by climate change, streams, especially in the alpine, will become shallower, with less flow, elevating water temperature and the risk to important aquatic species.	

Table ES1 (continued). High level summary condition statements by major stream vital sign category, Glacier National Park sentinel sites, 2007 to 2019. Hyperlinks connect to more detailed summary condition statements in the body of this report, which themselves synthesize the full analytical results. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value (i.e., increasing temperatures are usually signs of decreasing condition and the arrow thus would point downwards).

Vital sign	Summary condition	Status, trend, and confidence
Physical habitat	Physical habitat includes variables like channel shape and size, sediment levels, and riparian vegetation, and it is the template upon which all ecological processes in the park's streams occur. Stream habitat can be impacted by small scale disturbances from roads. It can also be impacted by large-scale processes like climate change that are impacting stressors like extensive forest fires and changing runoff patterns in watersheds. Overall, trends from 2007 to 2019 in the condition of physical habitat at Glacier sentinel sites showed few areas of concern. There was some indication of possible fine sediment issues and, using a conservative assessment point, the frequency of invasive/exotic plants was in a non-reference condition at many sites. Otherwise, the physical habitat at the sentinel sites over our period of record was generally stable and largely in reference condition.	
Bio-assessment	The macroinvertebrate and diatom communities of Glacier are some of the most diverse examples known in the world. They provide an excellent avenue for bioassessment. Bioassessment is the use of measures of a biological assemblage to assess and monitor natural systems. It is the primary way we address objectives focused on estimating status in the condition (or ecology integrity) of Glacier's sentinel stream sites. In this report we focus on stressor specific macroinvertebrate and diatom indices. It is useful to directly link macroinvertebrate response to important specific stressors in Glacier. This can be diagnostic and thus able to support more specific resource management needs. Future work will present an updated synthetic multimetric index for macroinvertebrates and diatoms. Overall, our bioindicators of specific stressors in the park suggested some caution, but in general, this is isolated, and bioassessment indicators overall were in a reference condition. Accounting for bioassessment indicator values sometimes improved the condition assessment for chemistry and habitat indicators. This is notable for fine sediments, where we do not see as strong of a biological response as we do in actual habitat data. In contrast, our bioindicator of water temperature suggests that warming water at our sentinel sites is impacting stream biota, once again showing the general importance of this climate driven process. There is also a fair degree of biological response to nutrients, yet when combined into the state's integrated nutrient protocol, we do not see nutrient impairment at any site over our period of record.	

Acknowledgments

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We would like to especially thank the staff at Glacier National Park and the Crown of the Continent Research Learning Center for accommodating and supporting logistics, housing, and use of office resources, and with helping us adapt network monitoring to the management needs for the park.

Abbreviations

AIC:	Akaike Information Criterion
ATtILA:	Analytical Tools Interface for Landscape Assessments
BPJ:	best professional judgment
CCE:	Crown of the Continent Ecosystem
CDF:	cumulative distribution function
CWA:	Clean Water Act
EMAP:	Environmental Monitoring and Assessment Program
EPA:	U.S. Environmental Protection Agency
FLBS:	Flathead Lake Biological Station
FSBI:	Fine Sediment Biotic Index
Glacier:	Glacier National Park
GMP:	General Management Plan (National Park Service)
GRSA:	Great Sand Dunes National Park and Preserve
GRTS:	Generalized Random-Tessellation Stratified spatially balanced survey design
HBI:	Hilsenhoff Biotic Index
HDI:	Human Disturbance Index
IMD:	Inventory and Monitoring Division
IPI:	Invasive Plant Index
ITIS:	Integrated Taxonomic Information System
LDC:	least disturbed condition
MDC:	minimally disturbed condition
Montana DEQ:	Montana Department of Environmental Quality
MMI:	Multimetric Index
MMIGd:	diatom multimetric index specific to Glacier National Park

MMIGm: macroinvertebrate multimetric index specific to Glacier National Park

NAWQA: National Water Quality Assessment

NHD: National Hydrography Database

NLCD: National Land Cover Database

NMDS: nonmetric multidimensional scaling

NPS: National Park Service

ORW: Outstanding Resource Water

OTU: operational taxonomic unit

PCB: polychlorinated biphenyls

PCoA: Principal Coordinates Analysis

QAPP: quality assurance performance plan

QAQC: quality assurance quality control

RBS: relative bed stability

RIVPACS: River Invertebrate Prediction and Classification System

ROS: Regression on Order model

SC: specific conductance

SEI: Stream Ecological Integrity

S:N: signal to noise ratio

SOP: standard operating procedure

TPI: Temperature Preference Index

TSS: total suspended sediment

USFWS: US Fish and Wildlife Service

USGS: US Geological Survey

1 Introduction

1.1 The National Park Service Inventory and Monitoring Program

The purpose of the National Park Service (NPS) Inventory and Monitoring (I&M) Program (Fancy et al. 2009) is to develop and provide scientifically sound information on the status and long-term trends in the composition, structure, and function of park ecosystems (i.e., ecological integrity). As part of the NPS's effort to improve park management through greater reliance on scientific knowledge, a primary role of the I&M Program is to collect, organize, and make available natural resource data and to contribute to NPS institutional knowledge by facilitating the transformation of data into information through analysis, synthesis, and modeling of specific key vital signs.

The Rocky Mountain Inventory and Monitoring Network (hereafter, network) and its partner parks and scientific collaborators identified 12 high-priority vital signs for focused, long-term monitoring. The vital signs include Wet and Dry Deposition; Weather and Climate; Water Chemistry; Surface Water Dynamics; Freshwater Communities; Invasive/Exotic Aquatic Biota; Groundwater Dynamics; Wetland Communities; Invasive/Exotic Plants; Vegetation Composition, Structure, and Soils; Focal Species (beaver, elk, grizzly bear, and Great Sand Dunes National Park endemic insects); and Landscape Dynamics (Britten et al. 2007).

Stream ecological integrity (SEI) monitoring in Glacier National Park (hereafter, Glacier) addresses five of the network's 12 high priority vital signs: Water Chemistry, Surface Water Dynamics, Freshwater Communities, Invasive/Exotic Aquatic Biota, and Invasive/Exotic Plants. Three other high priority vital signs are indirectly linked with the SEI monitoring protocol, including Landscape Dynamics, Wet and Dry Deposition, and Weather and Climate. Network stream monitoring in Glacier is a multifaceted ecological health approach guided by a detailed SEI protocol (Schweiger et al. 2015). The SEI protocol describes an integrated approach to understanding status and trends in stream ecological condition, capturing the strengths of both fixed-site, water chemistry-based approaches and probability surveys. The protocol combines periodic (5–10 years apart) parkwide surveys of streams with annual "sentinel site" monitoring at four select sites.

1.2 Purpose of report

This report summarizes SEI monitoring conducted from 2007 to 2019 at six hand-picked sentinel sites. Because this is the first trend report on SEI monitoring at Glacier, we include justification and some detail for key elements of the protocol—readers experienced in stream monitoring and assessment may only need to skim Chapters 1 (Introduction) and 2 (Methods). We present additional analyses focused on baseline parkwide status from 2007 to 2009 in Schweiger et al. (2018).

1.3 Navigating the report for different audiences

This report provides both high-level summaries and comprehensive detail to meet the needs of its two primary audiences: park resource managers and network or collaborator scientists. We provide high-level results in summary condition tables in the Executive Summary and near the start of each Results section. We include detailed explanation in the main narrative and appendices to document our approach, methodology, and results, with a focus on supporting future monitoring work.

1.4 Glacier National Park stream resources

Glacier is in the "Crown of the Continent" ecosystem in northwest Montana in the northern Rocky Mountains. The Canadian Provinces of Alberta and British Columbia border the park to the north and contain Glacier's sister park, Waterton Lakes National Park (Figure 1). Elevation ranges from 949 to 3,180 m, with the Continental Divide bisecting the park north to south. The park's landscape is characterized by steep, high relief mountains interspersed with glacially carved U-shaped valleys. Glacier contains an extensive network of streams, most of which are high gradient perennial streams (Mast 2007).

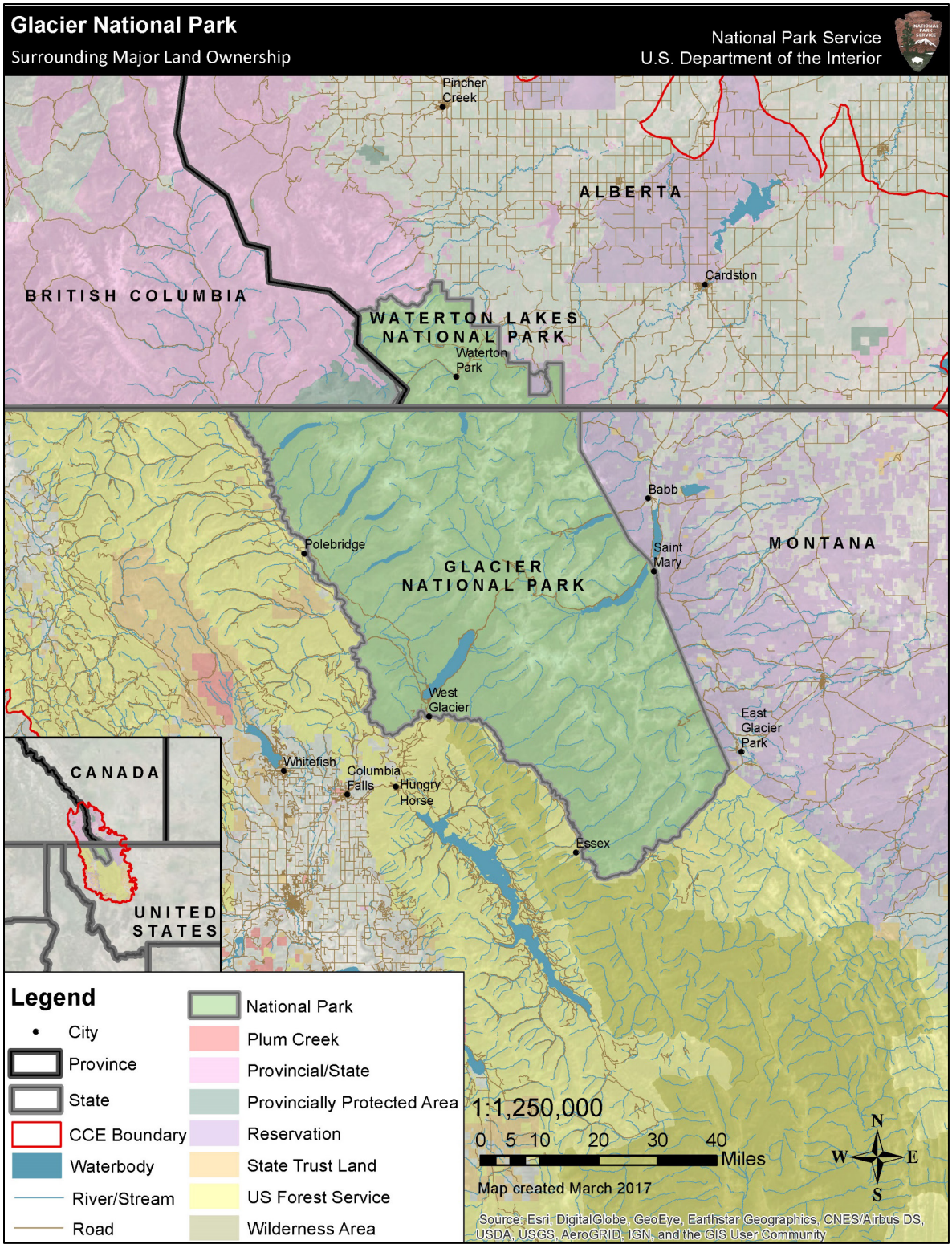


Figure 1. Major land ownership and other key features in and around Glacier National Park, Montana, USA, and Waterton Lakes National Park, Alberta, Canada. Inset shows location of the parks in the western United States and Canada and includes the boundary of the Crown of the Continent (CCE) region.

Glacier has been called the “water tower of the continent” (Hauer et al. 2016) because it is at the headwaters of three major river systems. Triple Divide Peak in northern Glacier occurs where the Continental and Hudson Divides converge. Waters flowing west enter the Columbia River and eventually the Pacific Ocean, waters flowing northeast enter the Saskatchewan River and the Arctic Ocean, and waters flowing east enter the Missouri River and Gulf of Mexico and ultimately the Atlantic Ocean. The North Fork and Middle Fork of the Flathead River form the western and southern boundary of Glacier. These large rivers are among a very few rivers in the conterminous 48 United States that are completely unregulated by dams or diversions.

Glacier has some of the cleanest water in the world and is treasured for its beauty, pristine condition, and ecological integrity (Figure 2; Long 2000. The World Wildlife Fund rates the biological distinctiveness of the Canadian Rockies Ecoregion that contains Glacier as “Continentally Outstanding” (Abell et al. 2000). Much of the Pacific Northwest region was heavily glaciated during the Pleistocene and, therefore, has few known endemic aquatic species. However, the Canadian Rockies Ecoregion served as a refugium for freshwater species during the last glaciation. As a result, Glacier’s streams, rivers, and lakes and associated groundwater and wetland systems support a wealth of biodiversity, from plants, including tiny diatoms (unicellular algae), to aquatic insects, to amphibians, fish, birds, and mammals. Over 500 taxa of diatoms have been recorded in the park, including several unique and new to science species (Bahls 2009). Aquatic insects are also diverse and include endemic threatened species such as the meltwater stonefly (*Lednia tumana*), which lives in alpine streams immediately downstream of glaciers and snowfields (Muhlfeld et al. 2011).



Figure 2. The North Fork of the Flathead River on the western border of Glacier National Park, 2009. (NPS / BILLY SCHWEIGER)

Glacier's streams, rivers, and lakes provide high-quality fish habitat inhabited by six native salmonids, including the threatened bull trout (*Salvelinus confluentus*) and the Montana Species of Concern westslope cutthroat trout (*Oncorhynchus clarkii lewisi*); both are indicators of high stream ecological integrity. Fourteen other fish species inhabit Glacier's aquatic habitats, including several sculpins (*Cottus* spp.), burbot (*Lota lota*), northern pike (*Esox lucius*; Downs et al. 2011), and several native catostomid species. The waters of the park also are home to five introduced nonnative salmonids, which affect native species and are of great concern to park management. Other aquatic or partially aquatic species of note include the Rocky Mountain tailed frog (*Ascaphus montanus*), common loon (*Gavia immer*), harlequin duck (Histrionicus histrionicus), American dipper (Cinclus mexicanus), northern water shrew (Sorex palustris), beaver (Castor canadensis), and the river otter (Lutra canadensis). Nearly every terrestrial species in the park interacts with streams, rivers, or lakes at some point in its life cycle.

1.5 Why monitor streams in Glacier National Park?

Streams are fundamental components of the ecological and cultural context of Glacier (Stottlemyer et al. 1997; Hauer et al. 2000; Seastedt et al. 2004; Mast et al. 2005; Hauer et al. 2007). They are often what visitors come to see, what visitors remember when they leave, and they have played an important role in the history (Figure 3) and culture of the park. Perhaps most importantly, the NPS recognizes that aquatic resources are some of the most critical and biologically important resources in the national park system and that they are vulnerable to degradation from activities both within and external to parks.

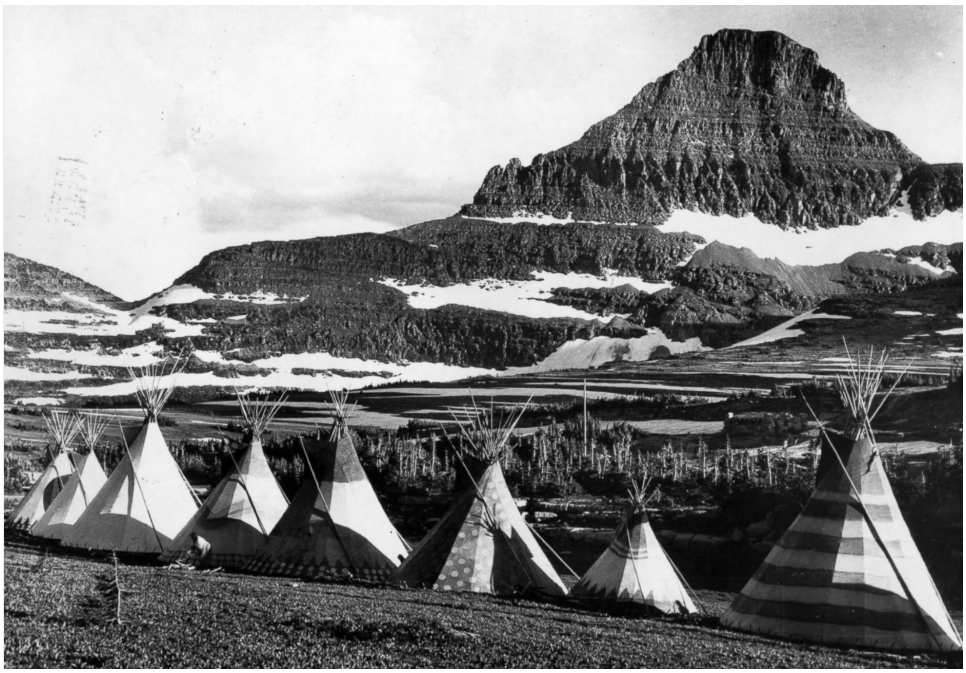


Figure 3. Blackfeet tepees at Logan Pass with Mt. Reynolds in background during the dedication ceremony for the opening of the Going-to-the-Sun Road, July 14, 1933. A Rocky Mountain Network stream sentinel site is now located on Reynolds Creek flowing just in front of the grove of trees visible in this scene. George A. Grant photo (Glacier National Park Digital Image Library).

1.5.1 Regulatory considerations

Regulatory considerations under the NPS Organic Act, the Clean Water Act (CWA), and other laws, regulations, and policies relative to network stream monitoring are summarized in the network SEI protocol (Schweiger et al. 2015) and updated in the survey report (Schweiger et al. 2018). While regulatory considerations provide important context, they do not drive SEI monitoring. Importantly, the park is the lead on any regulatory matter.

1.5.1.1 Montana non-degradation policy

One aspect of the regulatory environment relevant to SEI monitoring is the State of Montana's non-degradation policy for high-quality waters. This policy establishes that existing uses of high-quality state waters and the level of water quality necessary to protect those uses must be maintained and protected. The requirements for what constitute nonsignificant degradation and the conditions under which authorizations to degrade are allowed are described in Montana's *Nondegradation of Water Quality* policy (rule 17.30.701, Administrative Rules of Montana) . This policy provides protection for surface water in Glacier National Park by designating these waters as Outstanding Resource Waters (ORW) where *no degradation or permanent change in water quality from a new or increased point source discharge is allowed*. All state surface waters located wholly within the boundaries of designated national parks or wilderness areas as of October 1, 1995, are designated as ORW. SEI monitoring, under the guidance and direction of the park, should support the maintenance and protection of Outstanding Resource Waters in Glacier.

1.5.2 Climate change

Climate change has impacted and will likely continue to impact Glacier's aquatic resources (Hauer et al. 1997; Whitehead et al. 2009; Giersch et al. 2016; McKernan et al. 2018). In general, changes in climate will likely make it more difficult to achieve water chemistry standards in the park (Field et al. 2007). While there is regional variation, projected climate change effects across the West include loss of glaciers, less snow, earlier peak flows, less stream flow, warmer water temperatures, more frequent droughts, increased wildfire, and more intense storms. The interacting effects of climate change with other stressors, such as pollution, ultraviolet light, diseases, and environmental toxins are complex. Moreover, the ramification of changes in food-web structure and dynamics caused by the loss of some species and the expansion of others is unclear. Nevertheless, climate-induced changes in the hydrologic cycle and multiple proximate direct disturbances, such as elevated temperatures above species' physiological limits, often lead to changes in species distribution. This can include invasion by taxa that can threaten native or endemic species and the stability of aquatic ecosystems (Rahel and Olden 2008). The network considers the possible impacts of climate change on ecological integrity in the park's streams as one of the primary reasons why a comprehensive approach to long-term monitoring of streams is required, as this may be the best way to sort out any uncertainties associated with climate change.

1.5.3 Ecological integrity

Understanding and managing the streams and rivers of Glacier is best accomplished by treating them as complete hydrologic systems where natural processes are maintained, and human disturbances are minimized. Network monitoring aims to provide understanding of the hydrologic systems within network parks and the natural processes that streams and rivers support. While often short-term

regulatory considerations provide important context for SEI monitoring, our emphasis is on long-term ecological monitoring.

SEI monitoring focuses on the estimation and the interpretation of the ecological integrity of streams and rivers in the park (Schweiger et al. 2015). Ecological integrity is the capacity to support and maintain a balanced, integrated, and adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats of the region (Karr 1991). It is a complex, multidimensional concept and usually a single indicator is insufficient to characterize it. Therefore, we monitor an integrated set of stream characteristics, including benthic macroinvertebrates and algae, physical habitat (including stream hydrology), human disturbance, and chemistry. Our primary analytical focus is bioassessment of stream ecological integrity using macroinvertebrates and diatoms (Schweiger et al. 2015).

1.6 Monitoring objectives

The general goals for long-term ecological monitoring of Glacier streams focus on documenting the status and trend in stream ecological integrity, helping to understand the causes of changes, especially where a trend or change is relatable to human disturbance, and assisting in the application of SEI results and relevant ancillary information to park management (Schweiger et al. 2015, 2018). The following are SEI objectives focused on sentinel site status and trend:

1. Estimate status at sentinel stream sites in a suite of response measures.
2. Estimate long-term trend at sentinel stream sites in a suite of response measures.
3. Assess status and trend at sentinel stream sites through comparisons with assessment points of various types.
4. Relate temporal pattern in select stream responses to important ecological and anthropogenic drivers.

2 Methods

The following sections present brief summaries of how and why sites were selected, how data were collected in the field and/or harvested from ancillary sources, and how data were analyzed and interpreted. For additional content see Schweiger et al. (2015, 2018).

2.1 Sample design summary

Sample designs define locations in space for the collection of data, sample sizes required to meet monitoring objectives, and the strategy for arraying samples across time (Olsen et al. 1999). We employed a suite of three sample designs (Table 1): (1) "random" probabilistic surveys, (2) targeted hand-picked sentinel designs, and (3) targeted model-based gradient designs.

Table 1. Summary of sample designs and sample events from 2007 to 2019 in Glacier National Park stream ecological integrity (SEI) monitoring. Full events include all SEI components. Partial events only include chemistry and, starting in 2018 when safe, stream flow. Quality assurance and quality control (QAQC) events include duplicate or other-directed samples of specific elements of the SEI protocol. GRTS = generalized random-tessellation stratified spatially balanced design.

Sample design	Description
Survey sites	Forty-one sites sampled during the 2007–2009 survey 85 times with full events and four additional partial events. Over this period, we also conducted 27 unique QAQC events at survey sites. Sample design is a GRTS with unequal probability sampling across accessibility weighted stream order classes.
Sentinel sites	Six sentinel sites sampled three or four times per year. From 2007 to 2019, there were 63 full events at sentinel sites, with an additional 129 partial events. Over this period, we also conducted 69 unique QAQC events at sentinel sites Sample design is targeted.
Gradient sites	Six sites sampled once during the 2007–2009 survey. Sample design is targeted.

Table 2 lists the abbreviated names, site codes, and site descriptions for sentinel stream monitoring sites at Glacier National Park.

Table 2. Sentinel stream monitoring sites at Glacier National Park.

Site description	Abbreviated site name in this report	Site code
Swiftcurrent Creek on the east side of the park	Swiftcurrent Creek	GLAC_S01
Reynolds Creek at the top of Logan Pass	Reynolds Creek	GLAC_S04
Ole Creek in the southwestern corner of the park	Ole Creek	GLAC_S05
Middle Fork of the Flathead River near West Glacier	Middle Fork Flathead	GLAC_S06
North Fork of the Flathead River at Glacier Rim	North Fork Flathead—Glacier Rim	GLAC_013
North Fork of the Flathead River at the park border	North Fork Flathead—Border	GLAC_022

2.1.1 Survey

The probabilistic generalized random-tessellation stratified spatially balanced design (GRTS) or random survey of sites is a unique feature of the SEI protocol in Glacier. This random selection allows valid statistical inference to the population of streams in the park, with known confidence, from a relatively small sample. Figure 4 shows the location of all survey sites sampled in 2007–2009 in the park. Schweiger et al. (2015, 2018) present more detail on the survey design.

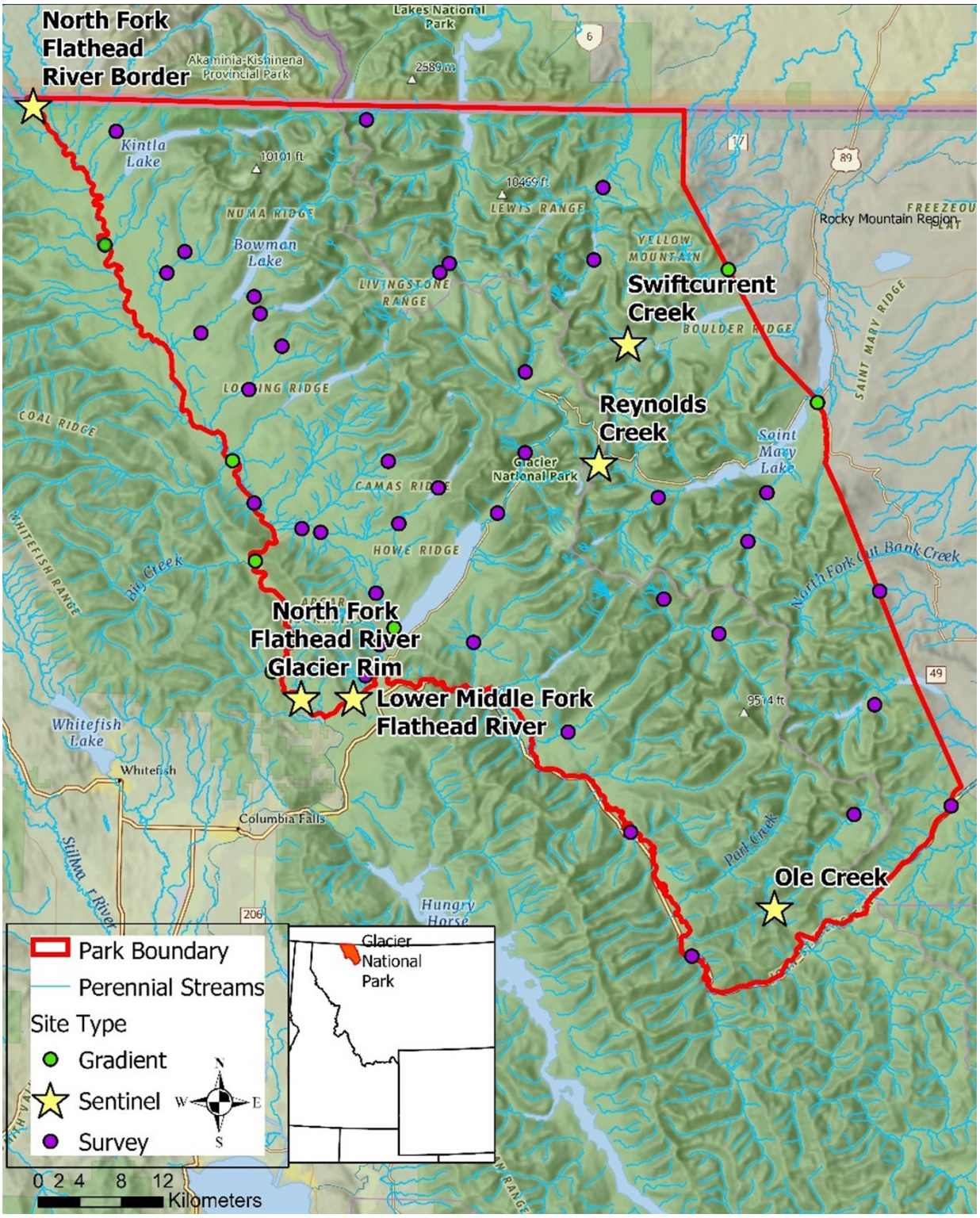


Figure 4. Stream sample sites in Glacier National Park. Inset shows the location of the park in Montana.

2.1.2 Sentinel

The bulk of the data in this report comes from sentinel sites. The most important distinguishing features of a sentinel design are (1) a judgment-based selection of stream site locations, (2) multiple samples at each site per year, and (3) no intention to make inference beyond the site or immediate stream reach without complex modeling. Sentinel sites (Figure 4) were located subjectively based on a suite of criteria developed by network staff, collaborators (especially scientists who have been working in Glacier streams for decades), and Glacier staff. Most importantly, sentinel sites were located based on an expectation that the sites are important and useful for understanding site specific patterns in stream condition over time. Sentinel sites were not selected with the expectation that they were in a reference state.

Sentinel sites in Glacier (Figure 5, Table 2) include the North Fork of the Flathead at the park border, Ole Creek in the southwestern corner of the park, Reynolds Creek at the top of Logan Pass, and Swiftcurrent Creek on the east side of the park. The North Fork and Swiftcurrent sites were selected because of the long-term data that are already being collected by the US Geological Survey (USGS), and because they each represent important stream and watershed types in the park. The Reynolds site was selected in part because it is one of the few accessible alpine streams in the park, but largely because of the important processes occurring in alpine streams in a landscape like Glacier National Park. Ole Creek was selected because it is a monitoring site for the Montana Department of Fish, Wildlife and Parks and is representative of an important middle elevation, high energy stream type in the park.

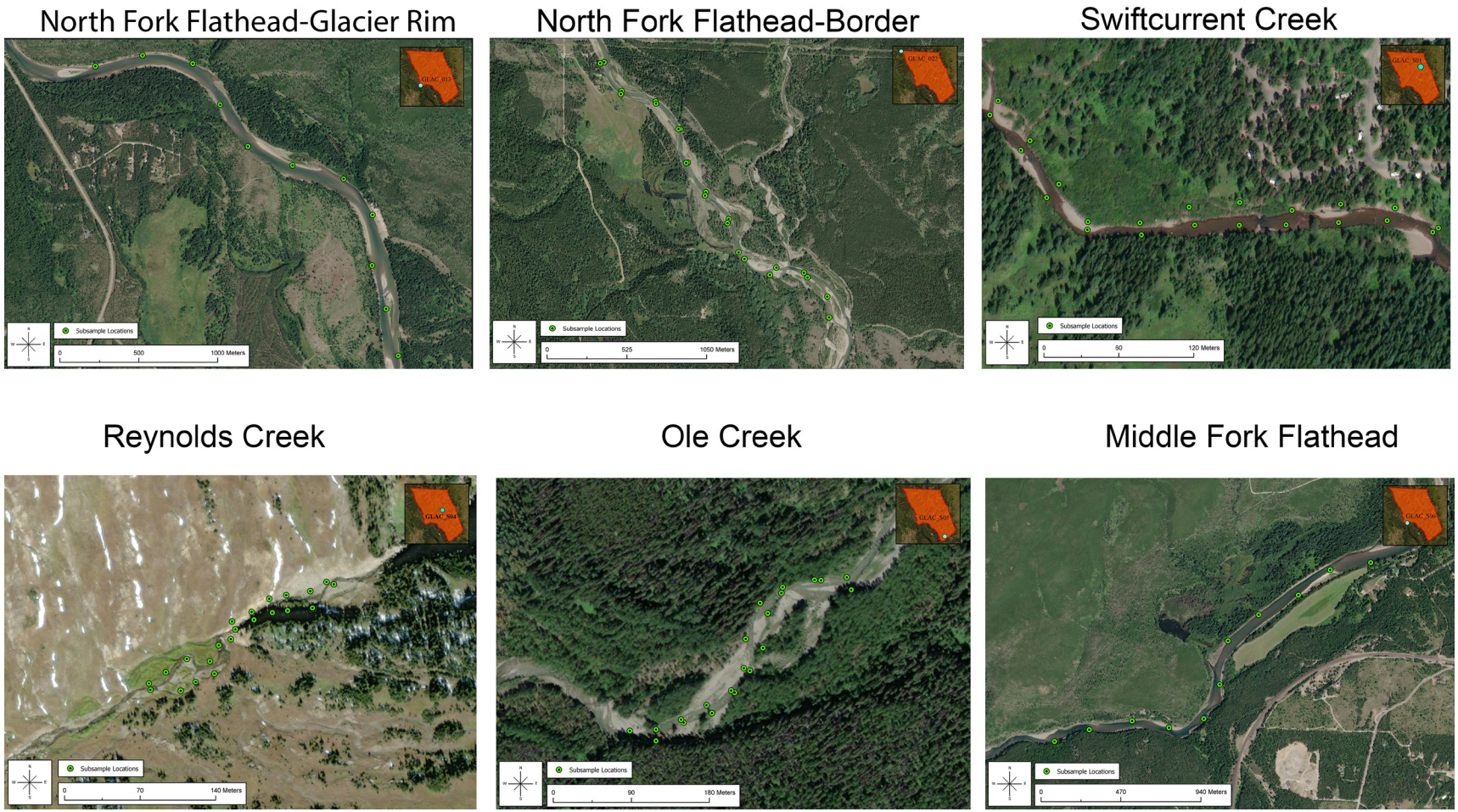


Figure 5. Reach scale maps for all six sentinel stream monitoring sites in Glacier National Park.

In 2018 we added two sentinel sites, one on the North Fork of the Flathead River at Glacier Rim and one on the Middle Fork of the Flathead River (Figure 5) at the request of Glacier National Park. Both sites are near where these two large rivers leave the park. They provide additional data on these important rivers that likely integrate and reflect conditions across their large watersheds. These areas also are used heavily by park visitors for recreation and are both designated Wild and Scenic Rivers.

2.1.3 Gradient

The gradient design is like the sentinel design (see above) in several aspects, including (1) an expert opinion or judgment-based selection of stream sites, and (2) no intention to make inference beyond the site. However, the criteria used to locate gradient sites differ from those used to identify sentinel sites. Data from gradient sites are primarily used to characterize stream condition at the extremes of anthropogenic and natural disturbance conditions that influence stream ecology. Therefore, sites are targeted to streams we expect to be under stress or in pristine condition relative to the complete population of streams in the park. Figure 4 shows the location of all six gradient sites sampled in 2007–2009 in the park. Schweiger et al. (2015, 2018) present more detail on the gradient design.

2.2 Field methods summary

Stream monitoring field methods are well established, and we draw upon this wealth of knowledge for the network SEI protocol. Sources of SEI methods include the EPA Environmental Monitoring and Assessment Program (EMAP; Stoddard et al. 2005), Flathead Lake Biological Station (FLBS; R. Hauer, Professor of Limnology, University of Montana, pers. comm. 2007), U.S. Forest Service (Heitke et al. 2008), NPS Water Resources Division (WRD; Irwin 2006), USGS long term water quality monitoring (e.g., Fitzpatrick et al. 1998), and the Montana Department of Environmental Quality (Montana DEQ) (Montana DEQ 2012a).

All indicators and field methods are described in detail in Schweiger et al. (2015, 2018) and summarized in (Table 3). The design of the sample reach for both wadeable and non-wadeable (boatable) streams (Figure 6) follows the EPA's EMAP site layout (Peck et al. 2006) as modified by the network SEI protocol (Schweiger et al. 2015). All laboratory methods follow standard protocols implemented by each individual lab and include detailed quality assurance and control methods. The SEI protocol includes a draft Quality Assurance Project Plan (QAPP) that summarizes laboratory methods and quality assurance and control aspects within SEI monitoring. Schweiger et al. (2015, 2018) provides more details and reports Method Detection Limit (MDL) by lab, year, and parameter. A quality assurance/quality control report is currently in preparation by Schweiger and others for SEI samples from all parks, 2007 to 2019.

Table 3. Summary of indicators and associated field data collected within the stream ecological integrity protocol at Glacier National Park.

Indicator class	Field data
Water chemistry	• In Situ temperature (synoptic and continuous), water; oxygen, dissolved; pH; specific conductance • Major ions (dissolved and total) alkalinity, calcium, chloride, fluoride, hardness, magnesium, potassium, silica, sodium, sulfate, total suspended sediments • Nutrients (dissolved and total, as applicable) ammonia, nitrite + nitrate, nitrite, nitrate, nitrogen, orthophosphate, phosphorous, carbon (organic), chlorophyll-a in algae; ash free dry mass in algae • Total metals in water and sediment aluminum, arsenic, barium, beryllium, cadmium, chromium, copper, iron, lead, manganese, selenium, zinc
Physical habitat	• Instantaneous discharge • Continuous discharge (from nearby USGS gauge if available) • Thalweg profile • Channel morphology • Woody debris tally • Channel dimensions • Substrate size quantification • Fish cover • Riparian vegetation cover and structure • Human disturbance
Macro-invertebrates	• Reach-wide (composited across transects) sample • Operational Taxonomic Unit (OTU) level identification for a target count of 600 individual specimens in a sample
Algae	• Reach-wide (composited across transects) sample • ~Species level identification for a target count of 600 individual specimens in a sample

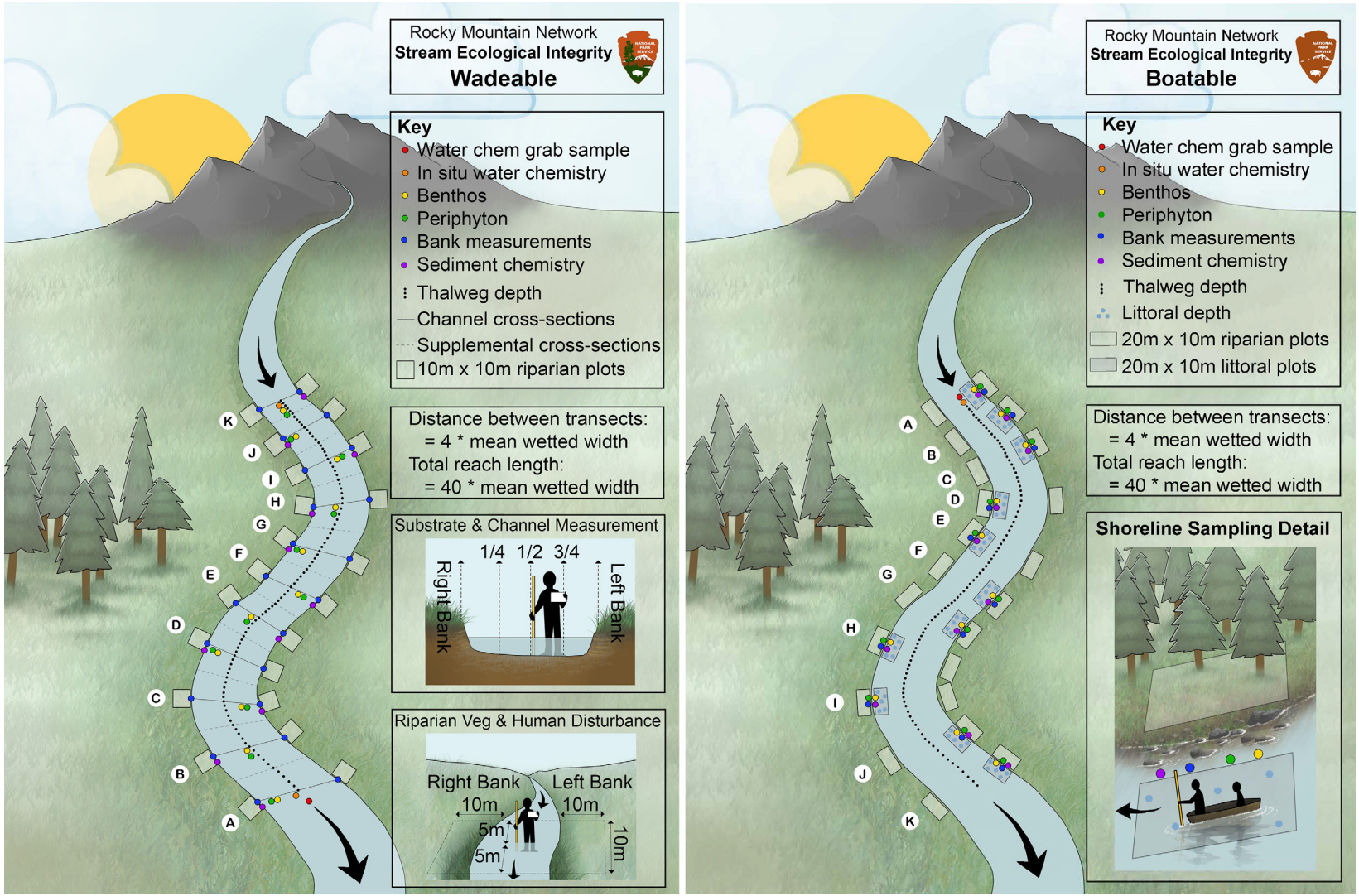


Figure 6. Stream ecological integrity sample stream reach plot in oblique profile: (left) wadeable, (right) boatable. Stream flow is from the top to bottom of the figure. Sub-sample locations, riparian plots, and the thalweg profile are all shown. Inset pictures show details for sampling.

2.2.1 Sample type and timing

Events at sample sites are of two types. First, full sample events typically include all methods (water/sediment chemistry, habitat, and biology) and are done at base flow, once per year (in Glacier, this is usually early to mid-August). Second, sentinel or partial sample events include only water/sediment chemistry and, when safe, stream flow. They are conducted on key limbs of the hydrograph (rising, falling, etc.). We generally conduct three sentinel (partial sample) events at each sentinel site per year (the main event at base flow also includes water/sediment chemistry). *Sample events are not timed to coincide* (*or avoid) storm events and our data should not be used to estimate the effect of storms.*

Full sample events vary based on the "wade-ability" of a site (the ability of crew to safely wade most of a sample reach) at the time of the sample, while sentinel events are identical regardless of the wadeable nature of a site. Non-wadeable sites require a non-powered raft to sample. Perhaps the biggest functional difference between the methods is that non-wadeable sites must be sampled in one direction (as rarely can the boat be rowed back upstream, especially in white water). This requires a few simplifications in the non-wadeable methods, and a few data types are collected remotely (i.e., channel slope and sinuosity come from a GIS) or taken from ancillary data sources (discharge is usually from USGS gauges that are often co-located with SEI sites on bigger rivers). However, in the end, data generated from the two types of events are nearly identical and allow nearly the same suite of metrics and types of analyses.

2.2.2 Vital signs (indicators and measurements)

The SEI protocol includes a broad spectrum of vital signs (i.e., indicators) to help evaluate the ecological integrity of streams in Glacier (Table 3). We include measures of both drivers and ecological responses, such that we may better understand the linkages between these and help the park apply network monitoring data to resource management. Core indicators include multiple measures of water and sediment chemistry (i.e., nutrients, metals, temperature); physical habitat (i.e., substrate composition, riparian cover, channel morphology ((and simpler aspects of fluvial morphology)), instantaneous stream discharge); and assays of two important biological assemblages, macroinvertebrates (Figure 7), and algae, including both soft bodied algae and diatoms, collectively referred to as periphyton. The biological assemblages provide an integrated perspective on ecological integrity because these organisms respond to environmental conditions over time and space. In addition, as part of the assessment of physical habitat, crews assess potential physical human disturbances (Table 3) and estimate their intensities largely based on proximity to the stream channel. These data are summarized into a c, see below). Many of the models we develop relate the gradient in HDI to synthetic bioassessment and physical habitat metrics.



Figure 7. Collecting macroinvertebrates on the North Fork of the Flathead, Glacier National Park, 2008. NPS / B. SCHWEIGER.

2.3 Ancillary data

We used several external (non-NPS) datasets to characterize and interpret data from sentinel stream sites in Glacier. External data were collected by state and federal agencies, academic collaborators, or national scale remotely sensed efforts (Table 4, see Appendix A for a map of external site locations).

Table 4. Summary of ancillary data used within the stream ecological integrity protocol at Glacier National Park. “GLAC GIS” refers to geospatial data managed by Glacier National Park.

Indicator Class	Indicator	Units	Source
Climate	Precipitation (annual, 30 year normal)	cm	DAYMET (Thornton et al. 2020)
	Ambient air temperature in catchment (annual, 30 year normal)	°C	DAYMET (Thornton et al. 2020)
	Growing degree days	Number of degree days over a base 3 °C	McMaster et al. (1997)
Water balance and water temperature	Runoff	mm	Lutz et al. (2010); Thoma et al. (2020)
	Climatic water deficit	mm	Lutz et al. (2010); Thoma et al. (2020)
	Soil moisture	mm	Lutz et al. (2010); Thoma et al. (2020)
	Evapotranspiration (actual and potential)	mm	Lutz et al. (2010); Thoma et al. (2020)
	Drainage	mm	Lutz et al. (2010); Thoma et al. (2020)
	Stream water temperature (reach scaled, modeled)	°C	Jones et al. (2014, 2017)
Geophysical	Stream slope	degrees	USGS (2017a)
	Percent of catchment as glacier	%	USGS (2017b), GLAC GIS*
	Percent of watershed burned over last 5 and last 25 years	%	GLAC GIS*
	Distance to upstream lake outlet	km	USGS (2017b), GLAC GIS*
	Watershed membership	categorical	GLAC GIS*
	Continental divide position	categorical	GLAC GIS*
	Elevation above mean sea level	m	USGS (2017a)
	Stream order (Strahler; at a map scale of 1:100k)	Index	USGS (2017b); Strahler (1952)
	Catchment area	m^2	USGS (2017a), GLAC GIS*
Human disturbance	Road density	length per unit area in catchment	US Census Bureau (2017), GLAC GIS*
	Road–stream crossing	number per unit area (km^2) in catchment	Riitters et al. (2000)
	Distance to nearest road	km	US Census Bureau (2017), GLAC GIS*

Table 4 (continued). Summary of ancillary data used within the stream ecological integrity protocol at Glacier National Park. "GLAC GIS" refers to geospatial data managed by Glacier National Park.

Indicator Class	Indicator	Units	Source
Human disturbance (continued)	Landcover/use	%	USGS (2017c); Hop et al. (2007)
	Structures	count per unit area	GLAC GIS*
Biological	Macroinvertebrate bioassessment metrics	Index	Stoddard et al. (2005); EPA (2016a); Montana DEQ (2009)
	Diatom bioassessment metrics	Index	Stoddard et al. (2005); EPA (2016a); Montana DEQ (2009)
Physical habitat	Habitat metrics	Indices or as detailed in the physical habitat section	Stoddard et al. (2005); EPA (2016a); Montana DEQ (2009)
Water chemistry	Nutrients, in situ, major ions	mg/l	Stoddard et al. (2005); EPA (2016a); Montana DEQ (2009)

2.3.1 Climate and water balance

Although temperature and precipitation are often used to characterize the climatic context of a study area, variables within the water cycle in a watershed (also known as water balance; WB) are often stronger indicators of ecological processes than temperature and precipitation alone (Dobrowski 2011; Munson et al. 2015; Thoma et al. 2020). Water balance variables (Table 5) reflect or incorporate the interaction of temperature and precipitation with topographic and edaphic (i.e., soil-related) properties that control temporal water availability in streams and rivers. For example, stream flow in much of the western North America is controlled by the abundance and timing of snow melt that occurs months after winter precipitation (Hostetler and Alder 2016); vegetation growth is closely linked to soil moisture and water deficit (Daubenmire 1968; Littell et al. 2008; Stephenson 1998), and fire is more closely related to aridity than precipitation (Littell et al. 2016; Williams et al. 2015), although wetting events diminish fire potential (Holden et al. 2018). Additional background on water balance is provided in Appendix B.

Table 5. Summary of water balance variables used within the stream ecological integrity protocol at Glacier National Park.

Water balance variable (units)	Description
Evapotranspiration (actual (AET) and potential (PET)) (mm)	ET is the movement (loss) of water from both plant transpiration and soil evapotranspiration PET is the "conceptual maximum" water loss from a large homogeneous vegetation-covered area that never lacks water and represents the climatic demand for water relative to the available energy AET is the realized loss of water from soil via evaporation and transpiration Our ET is used to estimate water demand and the water balance of an area, it is not intended as a localized ET (or "pan evaporation") often used in practical applications like irrigation scheduling or site-specific water resource management
Runoff (mm)	The amount of water that leaves a system exceeding water use by plants and storage in soils; it is estimated as the daily change in soil water minus AET plus direct runoff (i.e., from impervious surfaces)
Drainage (mm)	Runoff accumulated across days via base ("slow") and event based ("fast") runoff
Climatic water deficit (mm)	Unmet evaporative demand for water; what vegetation would use but does not have; estimated as PET minus AET
Soil moisture (mm)	Water stored in soil (mm) given idealized loss from soil via PET
Growing degree days (number of degree days over a base 3 °C)	Amount of energy available in a day for plant growth; estimates the accumulation of heat units required for plant growth and development or temperature above a base temperature threshold that is necessary for plant growth

2.4 Analyses

Schweiger et al. (2015, 2018) provides an overview of the general SEI analytical approach and Appendix B updates details for this report. We summarize key aspects of our analyses in the following sections.

2.4.1 Indices

We estimate several indices from SEI data. These include vital signs for human disturbance, macroinvertebrate and diatom assemblage structure, and physical habitat indices.

2.4.1.1 Human disturbance index

Human (and natural) stress is an important determinant of stream condition and indicators of these have long been included in stream monitoring and assessment. We develop a composite Human Disturbance Index (HDI) as a key part of SEI work in Glacier. Of note, the HDI is really a measure of human stress on a stream. Streams can respond to this stress, and this is really the measure of disturbance. Thinking of the HDI this way enhances the analysis and communication of the causal link from increased human population and use of wild area (as seen in increased Glacier visitation), to an increased HDI, to changes in physical and chemical habitat of a stream—finally resulting in changes in biotic integrity. The HDI provides an independent gradient against which measures of

stream condition and other SEI responses, such as physical habitat metrics, are assessed to determine how changes in disturbance regimes across the park or over time affect stream condition response.

Data estimating individual aspects of human disturbance are synthesized into the HDI, largely following the US Environmental Protection Agency (US EPA) (Stoddard et al. 2006, 2008) and other stream condition assessment programs (Montana DEQ 2012b). The following lists component metrics of the Human Disturbance Index (HDI) used within the stream ecological integrity protocol at Glacier National Park:

- Walls, dikes, revetments, riprap, or dams
- Buildings or other built structures
- Cleared lots (e.g., paved, graveled/dirt parking lot, foundations)
- Roads (paved or not) or railroads
- Inlet or outlet pipes
- Landfills or trash (e.g., cans, bottles, trash heaps)
- Parks or maintained lawns
- Row crops
- Pastures, rangeland, hay fields, or other evidence of livestock
- Logging
- Mining (including gravel mining)
- Park trails/campgrounds or other low-intensity facilities
- Density of road–stream crossings in site's watershed estimated using a GIS (Riitters et al. 2000)

These are mostly categorical measures that score the presence of various human disturbance categories. The likely influence is ranked based on spatial proximity to the stream channel, with features on the bank or in the channel weighted about twice as much as those outside the riparian corridor (Figure 8). Several metrics are also evaluated or confirmed in the office using a GIS, as GIS data will be more precise and accurate in some cases. The HDI is estimated by first calculating the number of riparian stations at which a particular type of disturbance was observed, weighting each observation according to its proximity to the stream, and then averaging over the riparian stations on the reach. At this state the index is equivalent to the "W1_HALL" index used by the US Environmental Monitoring and Assessment Program (EMAP) (Kaufmann et al. 1999) and with modifications by Montana DEQ (Suplee et al. 2005).



Figure 8. Land use on the Middle Fork Flathead sentinel stream monitoring site (GLAC_S06) in Glacier National Park. Aerial view shows adjacent hayfields, low intensity residential development, and, just upstream of the site, a golf course. The park is on the right bank or left side of this image. The presence of human land use features like these and their spatial relationship to the channel are incorporated in the Human Disturbance Index.

Schweiger et al. (2018) showed that the HDI, as originally conceived by the US EPA and MT DEQ, often did not include human disturbance spread across a broader scale, more distant from a site. In a wilderness context, where local disturbances can be rare, it was important to include these more remote potential impacts in the HDI. A large body of literature suggests that road density in a watershed is a useful proxy for large scale disturbances (i.e., Forman and Alexander 1998; Spellerberg 1998; Trombulak and Frissell 2000; McBride and Booth 2005; Theobald 2005). Therefore, Schweiger et al. (2018) modified the HDI to include a watershed scale road element derived from remotely sensed data, specifically a density of road–stream crossings (Riitters et al. 2000) per unit area in each site's catchment. This was derived using the park's road GIS data, augmented by US Census Bureau and Parks Canada road data for watersheds extending outside the park. A road–stream crossings metric is correlated with a simpler estimate of road density, but by focusing on actual crossings, it better isolates impacts resulting from interactions between roads and streams. The final HDI combines field-based disturbance data normalized to the highest HDI score recorded in the park by the network (Divide Creek in 2007) and the road–stream crossing density (importantly, the localized disturbance score and road density are thus given equal weight).

The road–stream crossing data used to augment the HDI is static, with only occasional updates that might include new road construction or other changes. The HDI includes a separate component

metric that also estimates road presence and potential impact to a site based on field observations. This road component metric likely captures new road construction visible from each site or easily inferred from local knowledge of the area near the stream or river. We include this road component as a stand-alone additional index of human disturbance in this report.

2.4.1.2 Bioassessment metrics

The primary way in which we analyze and interpret biological data is through bioassessment (Barbour et al. 2000). Bioassessment is the use of biologically based response measures to assess and monitor natural systems. We use the approach to estimate the status of stream ecological integrity (Schweiger et al. 2015, 2018). Bioassessment assumes that the composition of biological communities reflects the overall ecological integrity of a system. Evidence suggests it may detect stressors that other approaches fail to reveal and that it is an efficient, integrative way of conducting long-term monitoring (Karr and Dudley 1981; Karr and Chu 1997). Bioassessment has become a central element of most federal, state, and other monitoring programs and is also integral to monitoring programs worldwide, especially in Europe, Brazil, and several Asian countries.

2.4.1.3 Physical habitat metrics

Kaufmann et al. (1999) define physical habitat as structural attributes of streams that influence or provide sustenance to organisms within a stream. We exclude physical aspects of water quality, such as water clarity, temperature, and light intensity. We do include biological aspects, such as riparian vegetation and large woody debris in and near the channel, because of their role in modifying habitat structure. Many aspects of stream habitat can be both a response to external disturbances and a measure of stress on the ecological condition of streams. However, when not in a reference or long-term baseline state, we treat indicators of physical habitat as potential stressors on stream ecological condition. This is most obvious with our estimates of human disturbance, including the derived HDI summarized above.

Raw physical habitat data collected during an SEI sample event are summarized over all subsamples into multiple metrics of physical habitat. Many physical habitat metrics are synthetic, integrating diverse facets across space and time (e.g., Kaufmann et al. 2008). Using guidance in Kaufmann et al. (1999), Stoddard et al. (2005), and US Fish and Wildlife Service (USFWS 1998), we focus on 33 metrics in this report (selected from over 100 available metrics). These include summaries of areal cover estimates, more detailed and specific approaches for estimating woody debris, residual pool characteristics, sinuosity, and bed stability.

2.4.2 Status

We define status as a “snap shot” evaluation of an SEI response over a well-defined and relatively concise range in time (Olsen et al. 1999). Analyses presented in Schweiger et al. (2018) focus on estimating and understanding the population-scale status in important SEI responses in a baseline period of 2007–2009. Population-scale status results from Schweiger et al. (2018) are used in several ways in this report, most importantly as one form of baseline in assessments of how a response may have changed since 2007–2009.

Status may also be described at a single stream site, for example as the mean large woody debris density taken across transects or subsamples within a sample reach during a single or discrete number of sample events. Site-scale status cannot be easily or simply used for inference to unsampled locations (Olsen 1999). For example, if we were to estimate a vital sign response at a sentinel site, without complex models these results do not extend past the site itself (i.e., to a larger target population like the "streams of Glacier National Park" or even to the entire stream or river the site is on or to its catchment). It may be possible to make qualitative inference to unsampled streams in the park from specific sentinel site results, but this must be done with caution.

2.4.3 Trend

We define trend as a non-cyclic, directional change in a response measure that can be with or without pattern (Urquhart et al. 1998). Trend can be estimated at single sentinel sites through time, across sets of sentinel sites, or for extensive populations (i.e., "the streams of Glacier National Park") given appropriate randomized designs sampled across time (Larsen et al. 1995). When there is not sufficient data for trend estimates, we can provide baseline status estimates of change as the difference between two or a small number of time periods.

The general form of trend models used to analyze SEI data depends on several attributes of the data, especially the sampling interval or frequency of data collection. Our methods break into two forms: 1) data that are synoptic or sampled once to several times a year, and 2) data that are a time series or can be treated as continuous.

2.4.3.1 Responses with one to several replicates within a year: quantile regression

To estimate trend in responses collected one or more times a year we use quantile regression (QR: Koenker and Bassett 1978; Koenker 2021) to accommodate non-normal and heteroscedastic data. Several response datasets had extreme outliers, common in ecological data that QR effectively deals with. QR makes no assumptions about the distribution of the underlying data or residuals and is robust to outliers in the dependent variable. All models included a time predictor calculated as a sample date scaled and normalized to range between 0 and 1 (hence the name "Scaled Date"). We include additional predictors of stream flow, season, and lab or sonde type for chemistry responses. Human disturbance, habitat, and biology models did not include any predictor beyond scaled date. For non-climate responses we restricted models to estimating the conditional distribution at the median and assume all patterns were linear or could be approximated as linear. As our sample size grows, we will likely expand QR to include patterns in the tails (i.e., at the 5^{th} or 95^{th} percentile) and non-linear relationships. In this report, however, with sufficient sample size and an interest in extreme events, we do estimate 0.01, 0.5, and 0.99 quantiles in models of annual station based climate data.

To visualize and estimate trend for non-climate responses we use partial residuals from the QR models (Larsen and McCleary 1972; Breheny and Burchett 2017). A partial residual is a dependent variable corrected for all independent variables except a variable of interest in the model. For example, a trend coefficient (or slope) estimate from a partial residual removes the effects of season or flow, and importantly in chemistry responses, any differences across labs and their interaction with time.

Model results are presented in a familiar ANOVA format with coefficients, standard errors, and the significance of the term (p-value). While QR models have few assumptions about data, we do examine autocorrelation functions and qualify any results based on data with strong temporal structure. We report the overall adjusted model R^2 as estimated for a QR by Koenker and Machado (1999) and the change in AIC over a null intercept only model. We present QR analytical methods in detail in Appendix B. All model and data processing R code is available by request directly from B. Schweiger or the Rocky Mountain Network, via the NPS Integrated Resource Management Applications (IRMA) portal (https://irma.nps.gov/Portal/), and/or via github.

2.4.3.2 Continuous responses: General Additive Models of water temperature

Modeling time series can be tricky, as change over time in data that are continuous can be strongly non-linear and autocorrelated. Exploratory data analysis (EDA) of time series of water temperature suggested that General Additive Models (GAM) would be an ideal model form. Alternate methods such as linear regression have limitations with serially autocorrelated non-linear data and may therefore provide a poor description or prediction of pattern. We present our GAM analytical methods in detail in Appendix B as this is the first time they have been published by the network. All model and data processing R code is available by request directly from B. Schweiger or the Rocky Mountain Network, via IRMA, and/or via github.

To visualize final model results we again use plots of partial residuals. GAM results are presented in a familiar ANOVA format with parametric term coefficients, standard errors, the significance of the term (p-value), and relative importance (grouped for factor predictors like season). For smoothed terms we report the estimated degrees of freedom (EDF) that indicate the degree of smoothing in a fit, the F value, and significance of the smoothed fit (p value). We use p-values as a model selection criterion (in part); however, we do not require a predictor to be significant to be included if AIC is improved and the term is generally interpretable (*sensu* Murtaugh 2014). Finally, we report the overall adjusted model R^2, the change in AIC over a null intercept only model, and the form of any ARIMA term used. We used the mgcv GAM and GAMM models and the corARIMA function from the nlme library for ARIMAs (Wood 2011; Pinheiro and Bates 2000).

2.4.4 Statistical vs. ecological significance; how to interpret analytical results

For all models we treat statistical significance or ("p-values") as a somewhat fluid concept. Statistical significance is assigned at a p-value of ≤ 0.10. We believe a higher threshold (vs. 0.05) is appropriate for variable ecological data. We use the term "marginal significance" for p-values in the range of 0.10 to 0.20. By marginal we mean a result that is approaching our p-value threshold and thus with perhaps a few more data points or reduced variability in results might become truly significant. However, we will interpret patterns at even higher p-values if there are ecological reasons to do so. Finally, as many model forms estimate statistical significance via bootstrapping, the exact value of a significance test can vary slightly over repeated applications of a model. This supports not having stringent requirements as to the exact value needed for interpreting a result as meaningful.

Related to this is the interpretation of the direction in a trend. Numerically, this is the sign of the trend coefficient. Positive coefficients mean a result has increasing numeric values. However, this might mean that the inferred condition from this increase is moving in a negative direction—for

example, increasing arsenic concentrations probably means a reduced condition at a site. We attempt to be clear about these differences. Note that the direction of summary condition arrows is linked to condition, and so for our example here, the arsenic arrow would be down, indicating a worsening condition (even though arsenic concentration is increasing).

2.4.5 Baseline conditions as context for status and trend analyses

Assessing status and trends in stream condition requires baseline data. Our monitoring began in 2007 with a parkwide survey (Schweiger et al. 2018) that measured the indicators presented in this report with a statistical approach. To provide context for understanding how condition has changed at sentinel sites, we include a brief summary of baseline conditions in the results section for each metric.

2.5 Quality assurance and quality control

We report on extensive quality-assessment and control (QAQC) of all elements of SEI data used in this report in (in preparation by Schweiger and others). Quality-assessment and control is important for documenting the value of SEI data. *Quality assurance* (QA; Pirkey and Glodt 1998) refers to the overall management system, which includes organization, planning, data collection, quality control, documentation, evaluation, and reporting activities. QA procedures evaluate data for internal consistency and consistency over time. QA provides the information needed to ascertain the quality of data and whether it meets the requirements of the program. QA ensures that data will meet defined standards of quality with a stated level of confidence. *Quality control* (QC; Pirkey and Glodt 1998) refers to the routine technical activities whose overall purpose is error control. QC includes samples that quantify the bias and variability in measurements. Since errors can occur in either the field, the laboratory, or in the office, QC must be part of each of these functions.

2.5.1 Estimating censored data

A key piece of our QC efforts is to document and adjust for censored data. Censored data refers to values that are not precisely measured or reported because they fall below or above a detection limit of an analytical method. Across almost 32,973 records in the Rocky Mountain Network SEI database where we had detection limits, 36% of values were censored from a laboratory. Most of these were below a detection limit, with a smaller number present but below a quantitation limit or detected but not quantified. Metals in water or sediment had the highest rate, at 57%, with nutrients at 19% and major ions at 9%. These rates increase a bit if data are restricted to only Glacier sentinel sites (41% overall, 61% for metals, 29% for nutrients, and 12% for majors). The overall, and Glacier specific levels, of censoring are unsurprising given the dilute and oligotrophic water of most network streams. We used labs that had as low a detection limit for our parameters as possible, but with ours parks' typically "clean" water, some censoring is largely unavoidable. We apply standard imputation methods called Regression on Order Statistics (ROS; Lee and Heles 2005) and replacer censored data for all analyses. Censored values are by definition below a detection limit and detection limits are by definition smaller than detected values. Hence ROS replacement of censored data generates small imputed values, often near the minimum detection limit for a parameter. See Appendix B for more details on censored data.

2.6 Assessment

2.6.1 Overview

Assessment is the process by which the condition of an ecological resource, such as a single stream or a population of streams in a park, is placed into classes that describe or interpret this condition. In the most colloquial sense, these condition classes or these interpretations might be labeled "good" or "bad." In broad stroke, we compare our monitoring results to "assessment points" (AP, Bennetts et al. 2007, see below). We interpret these comparisons in a collaborative way with park staff and management in the context of ecological theory, NPS Inventory and Monitoring Division guidance, NPS resource management policies, and park management objectives. We use the terms "reference" and "non-reference" to describe the alternate states on either side of an AP. This is a simplification of the language used in Schweiger et al. (2018) and other network reports where we restrict "reference" to biological indicators following Stoddard et al. (2008) and used "negligible," "minor," "moderate," and "major" "disturbance" for chemistry and habitat indicators. Review from Glacier resource management staff suggested a single pair of terms—"reference" and "non-reference"—was preferable (with an "intermediate" designation as necessary). When a given result or set of results is in a non-reference state relative to a regulatory AP, we use the term "exceedance" (e.g., "10% of the results for total phosphorous at site GLAC_022 over our period of record exceeded the Montana DEQ criteria"). When a given result or set of results is in a non-reference state relative to any other type of AP (park directed, ecological, ecoregional), we use the term "outside" (e.g., "20% of the results for relative bed stability at Swiftcurrent Creek over our period of record were outside the ecoregional AP").

2.6.2 Assessment points

Assessment points are numeric values where scientists and managers have together agreed that they want to assess the status or trend of a resource relative to program goals, natural variation, or potential concerns. AP can have a basis in narrative statements, but we stress numeric over narrative AP to remove as much subjectivity from the process as possible. AP establish boundaries between alternate states or classes in measures of disturbance or ecological condition. Assessment points may or may not correspond to patterns in ecological processes—although it is preferred that assessment points have a known ecological interpretation. Even essentially arbitrary assessment points, if they are well defined, consistently applied, and explicit, still have value as a basis for comparisons across streams in space or a sentinel stream over time. AP values can define multiple levels in a disturbance or condition gradient. For example, the most severe AP might bound a non-reference state with a second value bounding intermediate condition. Network monitoring includes a variety of types of AP (Table 6, see Schweiger et al. 2015, 2018). These include Glacier resource management specified values, regulatory criteria, ecological thresholds, and values derived from reference sites in an ecoregion.

Table 6. Assessment point types, definitions, and examples as used in long-term stream monitoring in Glacier National Park. Assessment points are listed in order of their preferred application.

Assessment point type	Definition	Example
Defined by Glacier resource management	A value defined or adopted by Glacier National Park resource management staff.	Riparian Invasive Plant Frequency = 0
State / federal / tribal nation regulatory criteria	Criteria developed and defined by a state or federal agency with regulatory authority over a stream or river	Montana DEQ chronic total nitrogen summertime criteria for the Northern/Canadian Rockies ecoregion of 0.325 mg/L
Ecological thresholds	Threshold that defines a boundary between alternate ecological states given estimated or known relationships in an ecological process, such as critical habitat support	USFWS "Functioning Appropriately" for sustainable bull trout populations for 1.2 pieces of large wood per 100 m of stream
Ecoregional assessment points	95th percentile in a distribution of an indicator that increases with decreasing condition (like most water chemistry parameters as sampled from reference sites in the ecoregion(s) containing the park (see *Ecoregional assessment points* in Appendix A). Values larger than the 95th percentile are in a non-reference condition. For responses that decrease with decreasing condition, like most bioassessment indices, this changes to the 5th percentile, with non-reference below this.	A 5th percentile value of 2.78 for the Macroinvertebrate Fine Sediment Tolerance Index as estimated from 75 reference sites in the ecoregions Glacier is nested within

These assessment methods have been in use in one form or another for decades but were most explicitly defined in Stoddard et al. (2006). They are in use by the US EPA (EPA 2016a), and several states, including Montana (Suplee et al. 2005) and Colorado (CDPHE 2012). The Rocky Mountain Network uses a common assessment methodology across all our protocols in all six of our parks. We use terminology that aligns with earlier work by the NPS Natural Resource Condition Assessment program (https://www.nps.gov/orgs/1439/nrca.htm). We develop and present assessment point methodology for streams in Schweiger et al. (2018). Glacier National Park has accepted our approach as a general and flexible framework for interpreting SEI data in the park. We provide additional details and update our approach for application to Glacier National Park stream ecological integrity monitoring in Appendix A.

2.6.3 Summary condition

Select Glacier SEI results are summarized in "summary condition" statements within tables that include a brief description of the indicator, summaries of the status and trend in the condition of these indicators, and our confidence in the result. Summary condition statements are important for rapid integration into park resource management and interpretation. However, they are a simplification of details presented in our full results and discussion sections and should be used with caution.

2.6.4 Relationship to the regulatory authority of the Clean Water Act

Network stream monitoring is designed to document the ecological integrity or capacity of park streams to support and maintain a balanced, integrated, and adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats of the region. We believe this integrates well with the NPS Organic Act mission to conserve all resources "unimpaired for the enjoyment of future generations." To better interpret stream monitoring results, we utilize the framework and criteria developed by EPA and states under the CWA; however, our assessment is not formally tied to the CWA, nor do we use the term "impairment" as defined by the CWA or determine whether a "designated use" has been attained.

NPS does not have regulatory authority under the CWA over streams in parks because, in most cases, this authority has been delegated by the federal government to the state. Therefore, SEI assessment is conducted outside of State of Montana regulatory context. Interpretation of water chemistry standards or 303(d) impairment decisions are beyond the scope of our analyses and are not provided here. SEI data are nevertheless useful for understanding and evaluating whether park streams are in good condition or impaired either under the NPS Organic act or under the CWA. Any comparisons we make using SEI or partner data to state or federal regulatory criteria will not include a statement as to whether a beneficial designated use was attained—this may be added via consultation between the park and Montana DEQ at a later date. If requested and appropriate, the network will provide data and interpretation to Glacier National Park for this purpose, but regulatory authority resides in Montana DEQ and its interaction with Glacier National Park resource management.

2.7 Databases and data availability

2.7.1 Databases

2.7.1.1 Biology and physical habitat

The Rocky Mountain Network Streams master multi-year database 2007–current contains datasets by protocol components (e.g., biota, physical habitat) as well as site and event metadata that is collected in the field. The SEI SQL database is located on the SQL Server instance: INP2300SQL01\NTWK and is the "ROMN_SEI" database. A data entry front end application is used to enter raw SEI field sheet data to the SEI SQL server database.

2.7.1.2 Water chemistry

Water chemistry data are housed in the National Park Service Water Resource Division (WRD) EQuIS system. Certified data in the NPS Water Resources Division system are subsequently made available on the Water Quality Portal, which is a national water quality data repository cooperatively managed by the USGS, the US EPA, and the National Water Quality Monitoring Council (NWQMC). Instructions are available on IRMA for downloading all certified network water chemistry data from the Water Quality Portal.

2.7.1.3 Continuous logger data

All NPS continuous logger data are uploaded to the NPS Aquarius system managed by the WRD. The WRD Aquarius system is the long-term repository for continuous water quality and temperature data. Data quality control and review are performed via heads up visualization tools within Aquarius.

Certified continuous data are available for visualization and download on the NPS Continuous Water Data Aquarius web portal. Instructions to access network SEI continuous water quality and temperature data are available on the IRMA portal.

2.7.2 Data preparation

Water and sediment chemistry data used in this report were extracted from WRD EQuIS and prepared for analysis using R. The chemistry database used in all analyses used in this report contained a total of 32,973 records of 103 parameters across all network parks. Of these, 14,456 were from Glacier NP, with 10,697 from sentinel site events between 2007 and 2019. Event types in the network database include field measurements (3159 records), field blanks (337), field replicates (1607), lab blanks (83), matrix spikes (90), and routine or core environmental samples (27,697).

2.7.2.1 Development of final chemistry analysis data

The compilation of the chemistry dataset used for analyses included several steps after extraction from EQuIS. These steps are detailed in our QAQC report (in preparation by Schweiger and others). In summary, they include discovery of missing values, confirmation of lab source, collapsing parameters to a common name (i.e., if they were named differently across labs), correction for differences in units of measure, and transformations into common chemical forms. We identified extreme (3× the interquartile range) and outliers (1.5× the interquartile range)—these were flagged but usually retained for analysis. Results influenced by lab analytical issues (holding times or shipping temperature thresholds exceeded, broken bottle seals) were also flagged and, based on their individual details, retained or not. Problematic values that were so out of character for a parameter or site were followed up with lab contractors as possible and further flagged/removed as necessary.

2.7.3 Data packages

Data presented in this report and additional figures, tables, and model output are available from the network. Quality checked data packages are on the IRMA portal within the NPS DataStore.

All report documents and data packages have been bundled into a Glacier National Park Stream Ecological Integrity (SEI) 2007–2019 status and trend report project.

Data packages consisting of datasets, metadata, and dataset manifest lists by protocol components (e.g., biota, physical habitat, etc.) have been created as separate data store references. The Glacier National Park records in the data packages defined below were used in this report. Below is the list of individual protocol component data packages:

- Site Event Metadata — https://irma.nps.gov/DataStore/Reference/Profile/2286602
- Biota Periphyton/Diatom — https://irma.nps.gov/DataStore/Reference/Profile/2293341
- Biota Macroinvertebrates — https://irma.nps.gov/DataStore/Reference/Profile/2286764
- Continuous Water Temperature — https://irma.nps.gov/DataStore/Reference/Profile/2286688
- Physical Habitat — https://irma.nps.gov/DataStore/Reference/Profile/2293308
- Water and Sediment Chemistry Analysis Dataset — https://irma.nps.gov/DataStore/Reference/Profile/2299442

3 Results and Discussion

The following sections present results and interpretation of Glacier National Park stream data collected across six sentinel sites sampled three or four times per year from 2007 to 2019. We organize this section by response types, including climate, human disturbance, chemistry, habitat, and biological. Each section begins with a summary condition, where readers with a high-level interest in the park's streams might focus.

3.1 Climate

3.1.1 Introduction

The climate of Glacier National Park is characterized by a cool to cold season stretching through fall, winter, and spring, and summers with warm days and cool nights (Pederson et al. 2010). The Rocky Mountains are oriented north to south, perpendicular to the prevailing westerly winds in the park, creating extreme east–west climate gradients and large amounts of snow at high elevations (Davey et al. 2007). It is frequently 10 to 15 °F cooler at higher elevations (NPS 2006a). Glacier's western valleys generally receive more rainfall than its east side, where strong winds and sunny days predominate (NPS 2006a).

In the following sections, we present summaries to help understand how climate influences our stream data over several spatial and temporal scales. We also include future climate projections to allow park managers to consider the interaction between future climate conditions and the park's streams and rivers.

3.1.2 Summary condition

Summary condition statements condense climate results into concise overviews of key aspects of trends, assessment, and possible management applications or concerns (Table 7).

Table 7. Summary condition table for climate at Glacier National Park. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Climate	Temperature and precipitation	Climate and water balance play an important role in the ecology of Glacier National Park's streams and rivers. Understanding climatic patterns can help in interpretation of stream condition. Climate change continues to be and will likely become even more of an important driver of stream ecological integrity in Glacier. Overall, trends in climate are for warmer temperatures and flat to slightly increased precipitation. More variability in precipitation is also likely. Based on the historical and future climate expectations, seasonal changes will include warmer and slightly drier summers and warmer wetter winters. These changes will impact the hydrology of streams in the park, often with reduced overall flow and a shift towards earlier peak discharge. We currently lack numeric assessment points for climate, but given overwhelming evidence in the literature, we score climate as in a non-reference condition, with a decreasing trend—both of which are of significant concern. Given the complexity of the models and data behind this, and of real site-specific variability in climate driven processes in the park's streams and rivers, we give our assessment medium confidence.	
Climate	Water balance	Water balance incorporates the interaction of temperature and precipitation with vegetation, soil, and topology (i.e., a north vs. south facing slope effect), determining water availability to streams and rivers. Patterns in stream response to changing climate data reveal the importance of water balance. It is uncertain whether future precipitation will increase or decrease based on climate prediction models; however, water balance models indicate that warmer air temperatures will lead to less water available for surficial resources (e.g., streams and vegetation) in the future. There is variability in trends across east to west and elevational gradients, with complex ramifications on streams and other park resources, depending on their location. We currently lack numeric assessment points for climate, but given overwhelming evidence in the literature, we score water balance as in a non-reference condition, with a decreasing trend—both of which are of significant concern. Water balance models are complex, and we give our assessment medium confidence.	

3.1.3 Climate: station level

Because of a strong east to west and elevation gradient in the park, we summarize station data for three locations in the park. The west side is represented by the West Glacier SNOTEL station located

near headquarters. The Flattop Mountain SNOTEL site, located in the northcentral part of the park, best represents the high elevation areas, while the Many Glacier site, located in the southeast portion of the park represents east side climate. Raw station level data are daily and thus allow a rich context for trend modeling.

3.1.3.1 Summer air temperature

All three summer annual median air temperatures significantly increased from 1950 to 2022 (Figure 9, Figure 10, Figure 11, and Table 8). West Glacier median summer values increased by nearly 2 °C from around 14 °C in 1950 to around 16 °C in 2020. Flattop, in the park's alpine zone, also had hotter summer air temperatures, with medians significantly increasing from around 9 °C to 11 C, from 1980 to 2022. Finally, Many Glacier, also had increasing air temperatures with median values increasing around 2 °C from about 13 to 15 °C. Notably, extreme values at Many Glacier also had increasing trends over the 35-year period of record indicating that the warmest summers on the parks east side have become meaningfully hotter over the years.

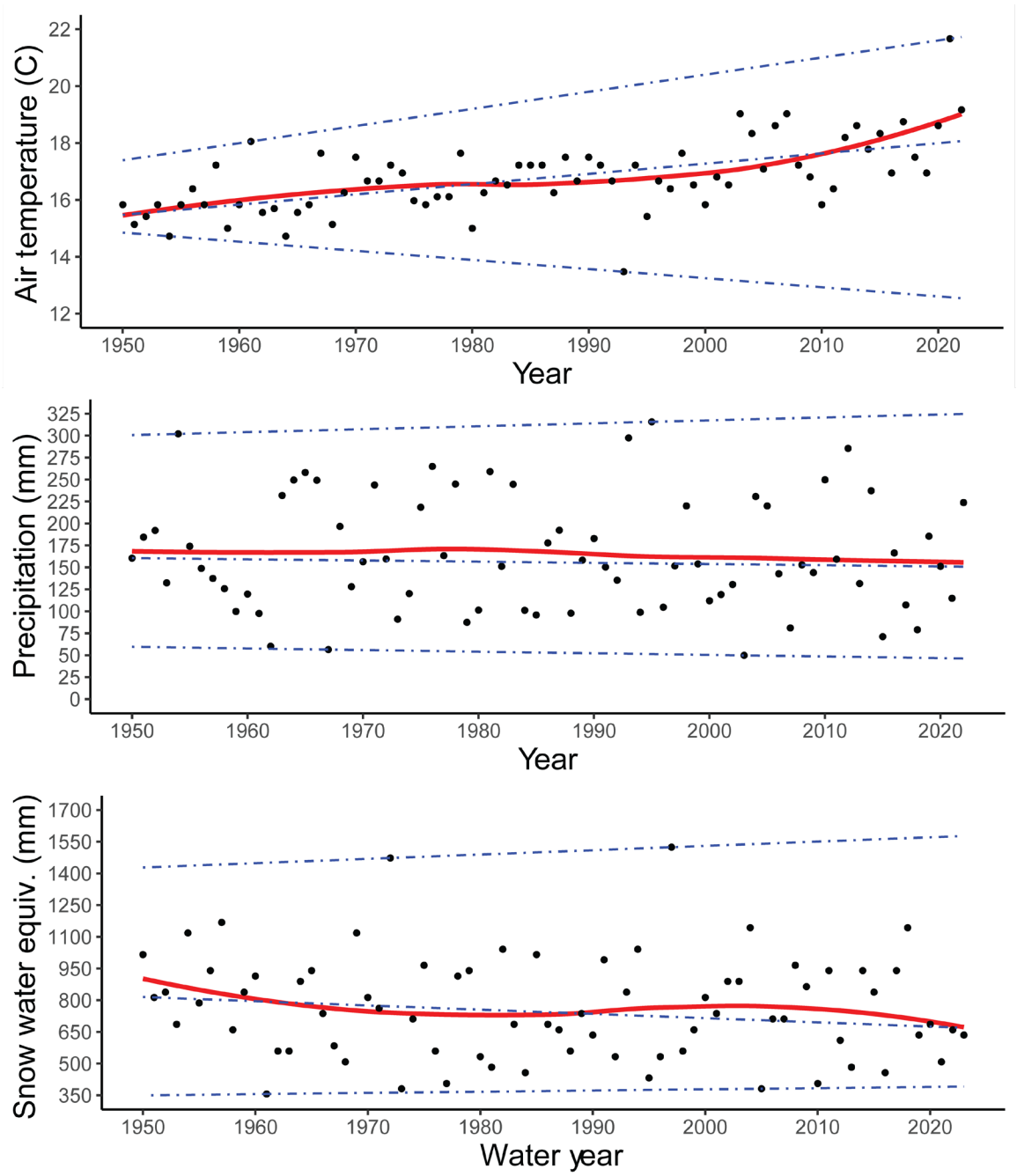


Figure 9. Climate at the West Glacier SNOTEL station in Glacier National Park, 1950 to 2022. Air temperature and precipitation data points are the median and total across daily June–August values, respectively. Snow water equivalent is the maximum value in each water year. In each panel the heavy red line is a loess smooth across median, total, or maximum values. Dot-dash blue lines are 0.01, 0.50 (median), and 0.99 percentile fits from quantile regression models.

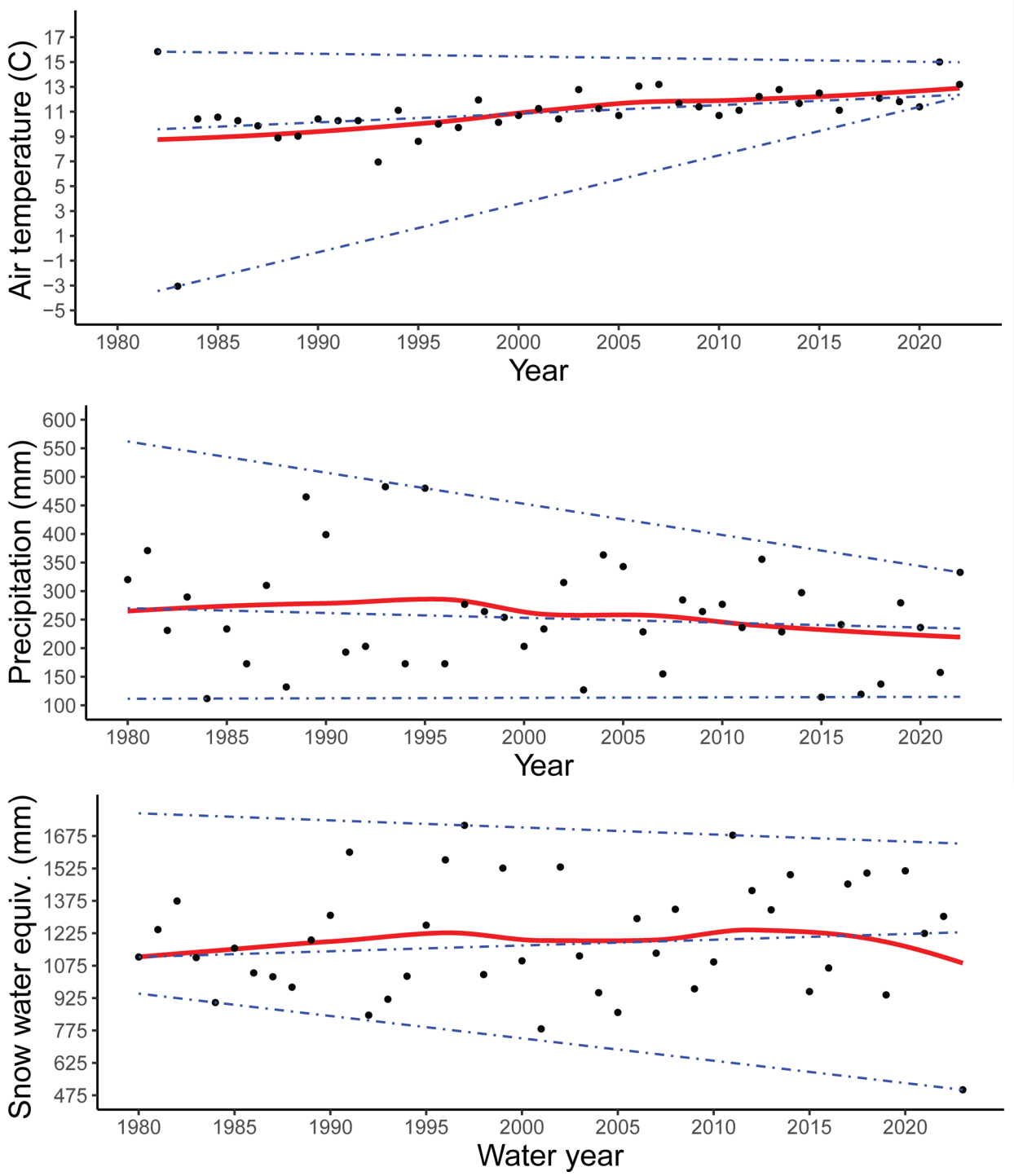


Figure 10. Climate at the Flattop Mountain SNOTEL station in Glacier National Park, 1980 to 2022. Air temperature and precipitation data points are the median and total across daily June–August values, respectively. Snow water equivalent is the maximum value in each water year. In each panel the heavy red line is a loess smooth across median, total, or maximum values. Dot-dash blue lines are 0.01, 0.50 (median), and 0.99 percentile fits from quantile regression models.

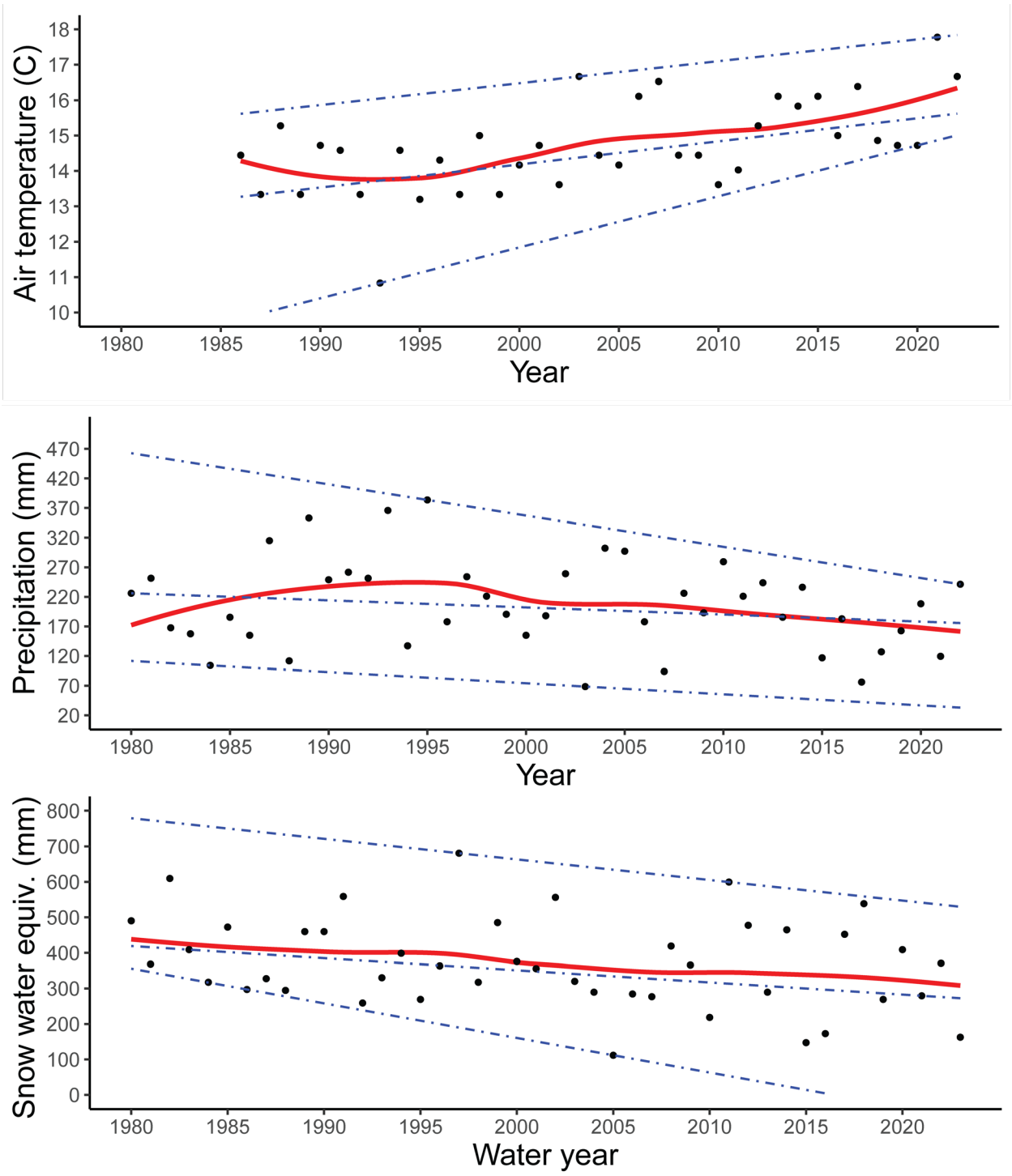


Figure 11. Climate at the Many Glacier SNOTEL station in Glacier National Park, 1980 to 2022. Air temperature and precipitation data points are the median and total across daily June–August values, respectively. Snow water equivalent is the maximum value in each water year. In each panel the heavy red line is a loess smooth across median, total, or maximum values. Dot-dash blue lines are 0.01, 0.50 (median), and 0.99 percentile fits from quantile regression models. Many Glacier SNOTEL air temperature data were not collected until 1986.

Table 8. Key results from quantile regression trend models of station level annual climate data at Glacier National Park. Note that periods of records vary (West Glacier: 1950 to 2022; Flattop Mtn and Many Glacier: ~1980 to 2022). Raw temperature and precipitation date are June–August daily data summarized to median and totals for each year, respectively. Snow water equivalent (SWE) raw data are maximum values for each water year. Results are shown for three quantiles: 0.01, 0.5 and 0.99. All models, except where noted with an asterisk (*), improved on null models, with a decrease in Akaike information criterion (AIC) ≥ 2 units. Decreasing trends have negative trend slope values and are in red. R^2 is shown for significant results only and is approximate, as described by Koenker and Machado (1999).

	Trend slope			Quantile significance			R^2		
Station-parameter	**0.01**	**0.5**	**0.99**	**0.01**	**0.5**	**0.99**	**0.01**	**0.5**	**0.99**
West Glacier median summer air temp. (C)	−0.03	0.036	0.06	0.29	**0.00** [a]	**0.00** [a]	–	0.2	0.5
West Glacier total summer precipitation (mm)	−0.18 *	−0.13 *	0.33	0.65	0.74	0.69	–	–	–
West Glacier maximum water year SWE (mm)	0.56	−1.99	2.04	0.58	0.13	0.74	–	–	–
Flattop Mtn median summer air temp. (C)	0.39	0.06	−0.02	**0.00** [a]	**0.00** [a]	0.80	0.66	0.19	–
Flattop Mtn total summer precipitation (mm)	0.08 *	−0.84 *	−5.45	0.94	0.57	**0.00** [a]	–	–	0.16
Flattop Mtn maximum water year SWE (mm)	−10.36	2.66 *	−3.26 *	0.16	0.56	0.53	–	–	–
Many Glacier median summer air temp. (C)	0.14	0.06	0.06	**0.01** [a]	**0.04** [a]	**0.08** [a]	0.57	0.08	0.46
Many Glacier total summer precipitation (mm)	−1.87 *	−1.19 *	−5.26	0.22	0.37	0.00	–	–	0.22
Many Glacier maximum water year SWE (mm)	−9.75	−3.81	−5.81	**0.00** [a]	**0.08** [a]	0.16	0.22	0.04	–

[a] Values are significant ($p \leq 0.10$) (also in bold font).

Absolute air temperature and change can lead to cascading effects on stream water temperature, runoff, and stream flow, which can have strong impacts on chemistry, habitat, and flora and fauna in streams. Patterns in the park are consistent with a large body of literature supporting increased air temperatures in the western U.S. For example, Monahan and Fisichelli (2014) show that the 10-year period from 2003 to 2012 had extreme values in minimum temperatures of the coldest month (i.e., warmer winters), mean temperatures of the wettest quarter (i.e., warmer springs), and the mean temperature of the coldest quarter (i.e., warmer winters). Perhaps most relevant to this report, many of the emergent results we see in our stream data may be connected to the warming air temperatures impacting the park.

3.1.3.2 Precipitation and snow water equivalent

Precipitation has strong interactions with drought, runoff, fire regimes, and water availability—all important in streams. Snow water equivalent (SWE) is a useful indicator of snowpack. Consistent with regional patterns (Pederson et al. 2010), trends in precipitation and SWE at our three stations are

more variable than air temperature. Neither had meaningful changes at West Glacier in median or extreme values. Flattop, in the alpine area of the park, had more variation in trends, with some extremes suggesting decreased precipitation or snowpack. At Many Glacier, however, median and the bottom 1% of SWE values decreased, perhaps driven by the elevated air temperatures on the park's east side. There were periodic droughts, especially from 2003 to 2011 in all locations, while from 2011 to 2018, there was higher than average precipitation. Similarly, Fisichelli et al. (2015) found high degrees of interannual or seasonal variability and no real trend in precipitation in Glacier.

3.1.4 Climate and water balance: watershed level

We extend the above station level patterns to watershed scale historical estimates as well as forecasts using two common human greenhouse gas emissions scenarios. We use the USGS National Climate Data Viewer (NCDV, Alder and Hostetler 2013) that summarizes climate across continuous grid surfaces for a variety of variables (e.g., temperature, precipitation, snow, runoff, etc.) through 2100. The National Park Service has moved toward regularly using the more severe (red lines in these figures) representative concentration pathway (RCP) 8.5 emissions model (see the Intergovernmental Panel on Climate Change for emissions models) for planning purposes (Gross et al. 2016; NPS 2021). We use the St. Mary watershed to represent the East side and the North Fork of the Flathead to represent the West side of the park. These two watersheds overlap with the station data presented above and with many of our stream sites.

3.1.4.1 Climate

Historical patterns in air temperature and precipitation in both the St. Mary and North Fork of the Flathead watersheds are like those seen in our station data (Figure 12) with clear, if small, increasing summer temperatures and generally flat precipitation from 1950 to 2005. Temperature trends appear to accelerate around 2005 through the end of our period of record in 2019, with a close visual match to our station data and trend models (Table 8). Forecasts for both watersheds show a clear warming trend expected through 2100 under both the RCP4.5 and RCP8.5 emissions scenarios, with the RCP8.5 increases more pronounced, at 4 to 5 °C; Figure 12). Warming is most pronounced in the winter and summer, with the largest changes from historical mean temperatures (Alder and Hostetler 2013). While more complex, variability in precipitation is expected to increase. Summer is the only season expected to see no change to a slight reduction in precipitation under the RCP8.5 scenario.

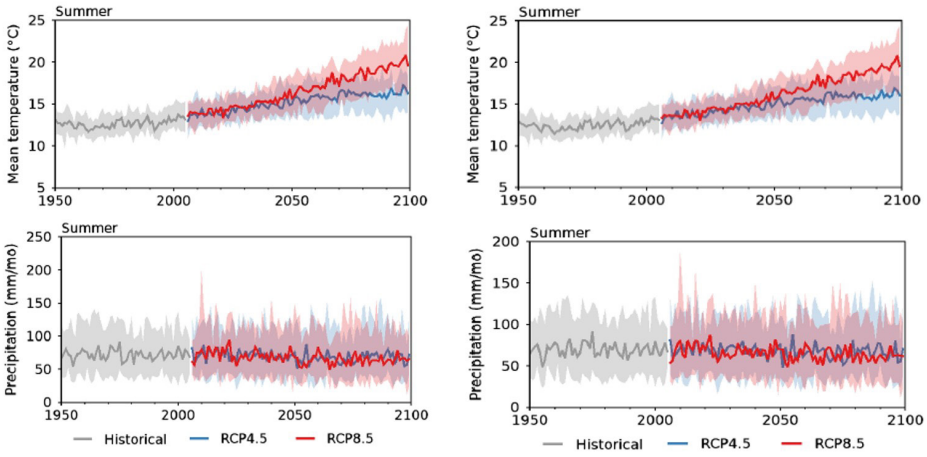


Figure 12. Summer mean temperature and total precipitation for the St. Mary and North Fork of the Flathead watersheds in Glacier National Park. Panels are as follows: air temperature (top left) St. Mary and (top right) west side North Fork of the Flathead watersheds; precipitation (bottom left) St. Mary and (bottom right) North Fork of the Flathead watersheds. In each, a historical period from 1950 to 2006 is in gray, with future predictions under representative concentration pathway (RCP) 4.5 (blue) and RCP8.5 (red) scenarios from 2006 to 2100. The median is indicated by the solid lines, and the 10th to 90th percentile range is indicated by the respective shaded envelopes. Figures courtesy Alder and Hostetler (2013).

3.1.4.2 Water balance: runoff and climatic deficit

Alder and Hostetler (2013) also provides historical and projected water balance models that incorporate the interaction of temperature and precipitation with topographic and soil-related properties that control temporal water availability in streams. We focus on runoff as a direct input from the water cycle in a watershed to stream flow, and climatic deficit, with its connection to drought and reduced stream flows (Figure 13). These water balance variables at the watershed scale are conceptually like our more nuanced and sentinel site specific treatment of them in the water temperature section below.

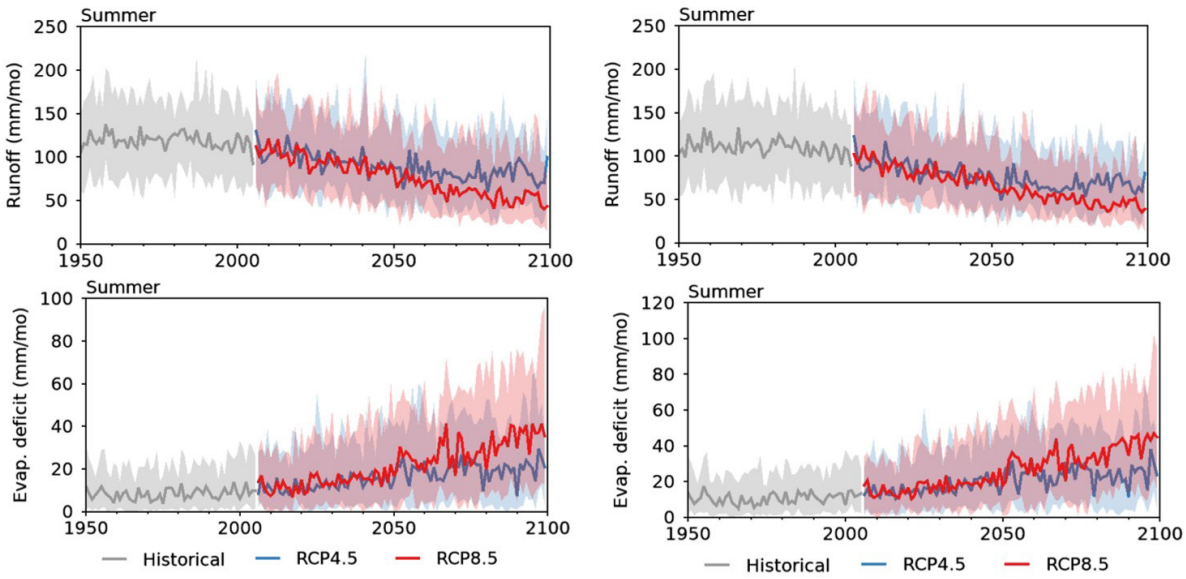


Figure 13. Summer (Jun–Aug) mean runoff (mm/month) and climatic deficit (mm/month) for the east side of the St. Mary Lake and North Fork of the Flathead River watersheds in Glacier National Park. Panels are as follows: runoff (top left) St. Mary and (top right) west side North Fork of the Flathead; climatic deficit (bottom left) St. Mary and (bottom right) North Fork of the Flathead watersheds. In each, a historical period from 1950 to 2006 is in gray, with future predictions under representative concentration pathway (RCP)4.5 (blue) and RCP8.5 (red) scenarios from 2006 to 2100. The median of 20 CMIP5 (Coupled Model Intercomparison Project Phase 5) models is indicated by the solid lines and the ensemble 10th to 90th percentile range is indicated by the respective shaded envelopes. Figures courtesy Alder and Hostetler (2013).

Runoff forecasts for both the St. Mary Lake and North Fork of the Flathead River watersheds show a reduction or negative trend through 2100 under both the RCP4.5 and RCP8.5 emissions scenarios. These extend and accelerate similar (but more subtle) reductions in runoff over the historical period. The accelerated reduction in runoff begins after 2005 and thus overlaps our stream monitoring data. Reduced summer runoff is balanced by increasing runoff in the spring and winter (Alder and Hostetler 2013) such that the annual trend is flat. However, these changes in the timing of runoff can have as large of a negative impact on streams as the net annual summer value. There are also pronounced differences across the park, with a strong elevation signal (Tercek et al. 2021; Figure 14), with lower summer runoff in alpine areas and less of a change in the lower elevation large river floodplains on the park's west side.

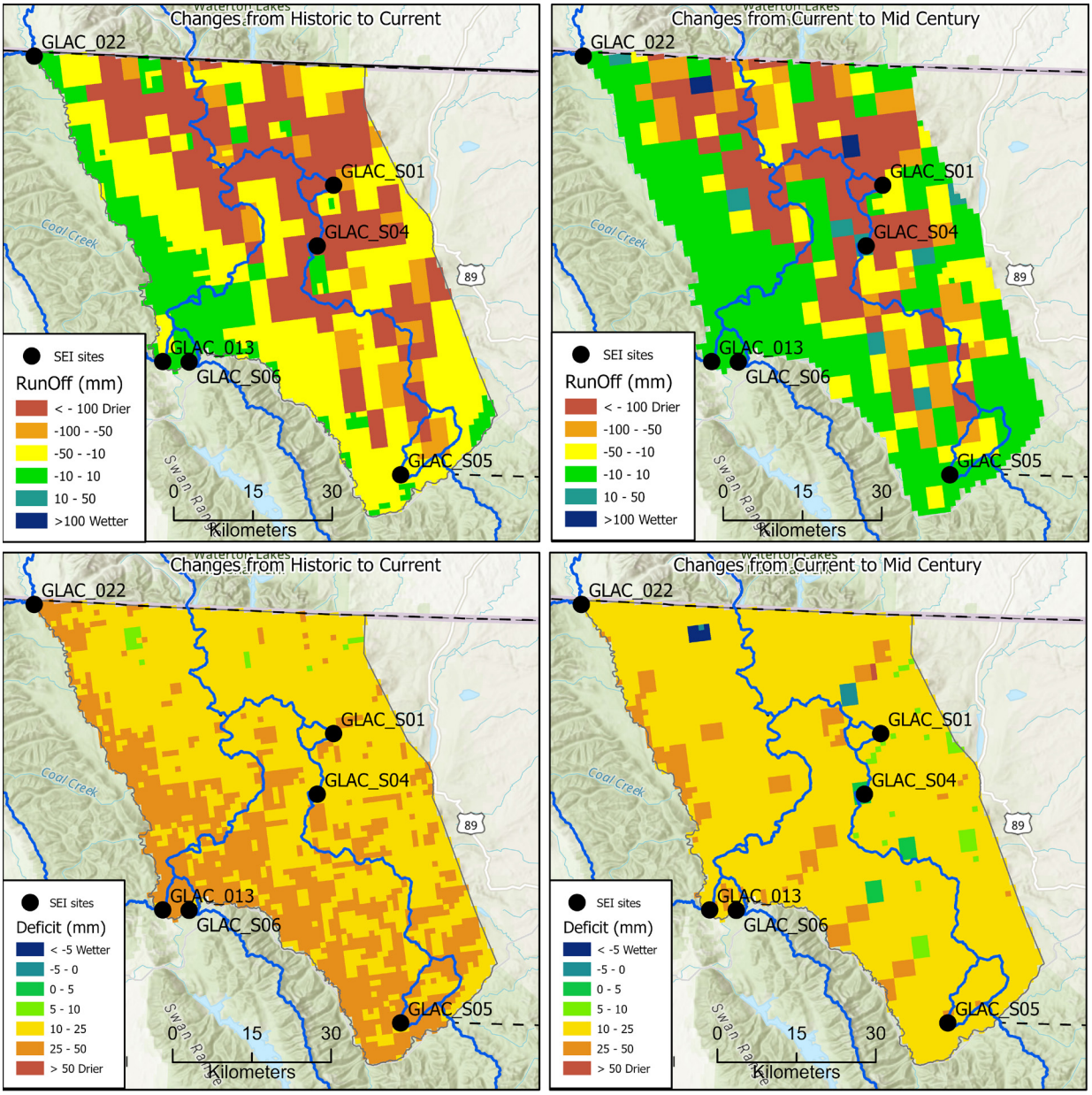


Figure 14. Differences between historical (1980–1999), current (2000–2019), and future (2040–2069) summer mean runoff (mm) and climatic deficit (mm) in Glacier National Park. Panels are as follows: top left = current runoff versus historical runoff; top right = current runoff versus future runoff; bottom left = current deficit versus historical deficit; bottom right = current deficit versus future deficit. Grid cell values for each period are averaged across each annual summer value. Future projections use representative concentration pathway (RCP) 8.5. Note that the color ramp scales differ in each panel, but colors are consistent with reds for increased dryness and blues for increased wetness.

Patterns in climatic deficit (i.e., drought) are essentially the inverse of the those for runoff, with increasing summer drought as climate change progresses in both the St. Mary Lake and North Fork of the Flathead River watersheds. However, across the park, elevation has less of an effect. Alpine areas retain some protection from drought moving forward even under the RCP 8.5 emissions

scenarios, with higher precipitation (mostly as snow) acting as a buffer against more severe deficit. Higher elevations will likely experience more variability in water balance over the years, with a cascade of effects on the hydrology, chemistry, water temperature, and other responses in alpine streams (many of which we detail in the results of this report).

3.1.5 Discussion and management application

Overall, these patterns reveal the importance of water balance. Precipitation model projections show a wide range of scenarios for the future, including potential for increases or decreases depending on the model. However, coupled with current and expected future warming, water balance models suggest that water available for resources will likely decrease over time.

The amount, timing, and duration of flow in the park's streams and rivers is a useful example of the management relevance of climate change. Even streams with a strong groundwater contribution are ultimately linked to climate and the water cycle—all water comes from precipitation as mediated by the water cycle and groundwater processes. This is a consistent element of the trend models or interpretations of status and trend we present in this report. Changes in climate and the water cycle will therefore impact the hydrology of streams in the park. Figure 15 gives one example of this for the Middle Fork Flathead (at our sentinel site GLAC_S06; Hamman et al. 2018), showing decreased flows and an earlier peak as climate change unfolds. Reduced overall flow and a shift towards earlier peak discharge as climate change progresses is a fairly common pattern in high elevation undisturbed watersheds typical of Glacier. The park has and will need to continue to develop adaptive management strategies to deal with these imminent changes.

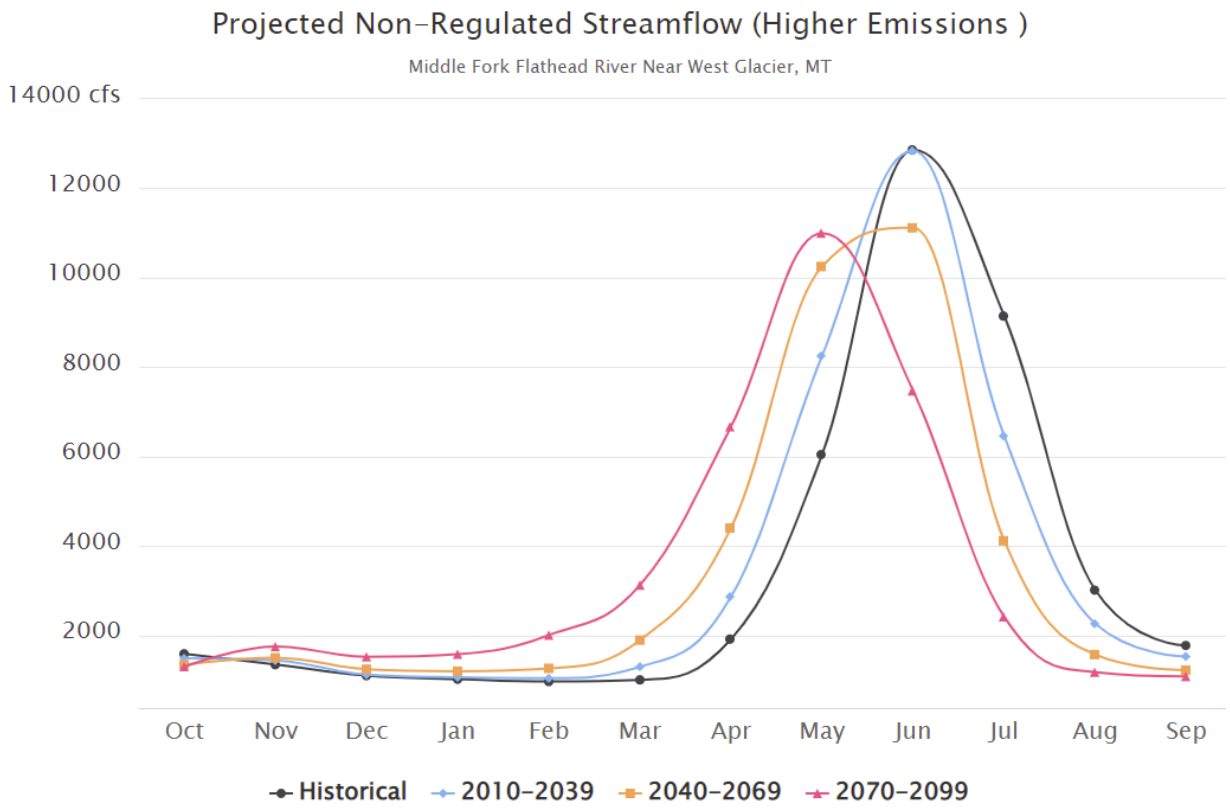


Figure 15. Annual hydrograph of the Middle Fork Flathead sentinel site over four periods (historical, 2020–2039, 2040–2069 and 2070–2099) under a representative concentration pathway (RCP) 8.5 emissions scenario and averaged across 10 climate projection models. Figure was estimated using the Variable Infiltration Capacity model (Hamman et al. 2018).

3.2 Human disturbance

3.2.1 Introduction

Streams and rivers can be sensitive to human disturbance and there is a long history of research seeking to understand these effects. Management of streams likely benefits from an understanding of these relationships, even in relatively undisturbed landscapes like Glacier National Park. Within the SEI protocol, we estimate a general site-scale index of disturbance during each sample event and use additional remotely sensed data to capture disturbances across larger scales. Estimating human disturbance and its role in influencing stream condition can be challenging in a wilderness park like Glacier—disturbances are often subtle and must be inferred from relatively simple data. Therefore, our estimates of disturbance are used as a relative index across sites and across time within a site, with interpretation scaled to Glacier only.

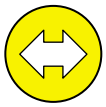
We estimate human disturbance with a Human Disturbance Index (HDI). The HDI is derived from both site level data collected as part of the physical habitat component of the SEI protocol and remotely sensed road data. These two scales more effectively capture human disturbances in the wilderness landscape of the park. Given the importance of roads as a general indicator of human disturbance, we also present a field level metric focused on crews' perception of road proximity and impact at each site.

The HDI provides an independent gradient against which measures of stream condition and other SEI responses, such as physical habitat metrics, are assessed to determine how changes in disturbance regimes across the park or over time affect stream condition response. It is a central element of our bioassessment models that estimate stream condition by relating stream biological assemblage structure to a gradient of human disturbance in and around each sentinel stream.

3.2.2 Summary condition

Summary condition statements condense human disturbance results into concise overviews of key aspects of trends, assessment, and possible management applications or concerns (Table 9).

Table 9. Summary condition table for human disturbance at Glacier National Park sentinel sites in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Human disturbance	Human Disturbance Index	Streams and rivers can be sensitive to local and large scale anthropogenic disturbance. Between 2008 and 2015, Glacier National Park averaged 2.12 million visitors per year. This increased to 3.09 million visitors per year between 2016 and 2019. Visitor impacts on streams can result from increased use of official park trails and increased social trail density, vegetation trampling, erosion of stream banks, deposition from vehicle exhaust, and human waste. Larger scale impacts such as high road density or extractive land uses have a well-documented impact on aquatic resources and can influence park streams even when these are outside the park. Management and interpretation of streams in Glacier likely benefit from an understanding of human disturbance, even in the park's relatively undisturbed landscape. SEI monitoring includes estimates of potential human disturbance(s) in and around a stream, summarized as a Human Disturbance Index (HDI) that combines multiple classes of disturbances at a site with watershed scale road density. There are no relevant regulatory criteria for the HDI, nor any known ecological thresholds. We therefore estimate ecoregional assessment points. No sentinel site HDI value was outside the assessment point over our period of record, suggesting sites were in a reference condition relative to human disturbance. Nevertheless, five of the six Rocky Mountain Network sentinel sites are in the front country, or immediately adjacent to front country, and have measurable human disturbance that may warrant some concern and future monitoring. To be clear, this result is specific to these sites, and we saw no indication of pervasive parkwide human disturbances to streams in our baseline data and do not expect this has changed since 2009. There were no significant or marginally significant trends in HDI at any sentinel site over our period of record. In other words, there is some evidence of disturbance at sentinel sites, but this remained constant—generally undisturbed from 2007 to 2019. However, trend in HDI at the Swiftcurrent Creek site approached significance, increasing over time. This generally matches increased visitation in the area over this period. We also saw increases in the field-data based road component of the HDI at the North Fork Flathead—Border and Swiftcurrent Creek from 2007 to 2019. At the North Fork site, this matches increased road construction evident in aerial imagery of this region of the park. Confidence in HDI data and its assessment is medium, given complex scale effects in these relationships.	

3.2.3 Assessment points

There are no relevant regulatory criteria for the HDI nor any known ecological thresholds. We therefore estimated ecoregional assessment points for both HDI and the road component metric using ecoregional reference sites. Note that for these two metrics, reference sites were limited to a subset of sites in Glacier National Park determined to be in a reference condition. Schweiger et al. (2018) and Appendix A provide details on how park reference sites were defined following methods from Montana DEQ (Suplee et al. 2005). The assessment points for HDI and roads were defined at the 95th percentile in the distributions of data across park reference sites (or HDI = 0.85 and Roads = 0.5). The HDI index ranges from 0 to 1, with higher scores indicating more human stress. Values below these assessment points are in the reference range3.2.4 Parkwide 2007–2009 baseline

As discussed in the Methods (section 2.4.5), we begin each metric's status and trend analyses with a brief summary of baseline conditions for context.

The HDI and the roads component metric in the 2007–2009 baseline survey both had low scores. Twenty-two stream sites out of 49 had an HDI score of 0. Of the other 27 sites, most (21) had minor or moderate human disturbance and only 6 sites had any likely meaningful human disturbance (Figure 16). HDI had a population scale median score of 0 and a mean of 0.17. A relatively small percentage of the park's streams (11.4%) had HDI or road scores above the assessment point or in a non-reference state. This shows that measurable human disturbance of streams at Glacier is restricted to front country locations.



Figure 16. Divide Creek gradient site (GLAC_G01) in Glacier National Park in the east side's main visitor support area. This site was listed by the State of Montana as impaired from sediment in 2007 (it is now delisted). This site had the highest Human Disturbance Index (HDI) score in the 2007–2009 baseline survey and therefore "sets the scale" for all other HDI values.

3.2.4 Condition

Neither the HDI nor the road component metric was outside the assessment point at any site over our period of record, suggesting these sites were all in a reference condition relative to human disturbance. Five of the six Rocky Mountain Network sentinel sites located in the front country or immediately adjacent to front country and had measurable non-zero median human disturbance.

3.2.5 Trend

There were no significant or marginally significant trends in HDI at any sentinel site (Table 10, Figure 17) over our period of record. However, trend in HDI at Swiftcurrent Creek approached

significance ($p = 0.13$), suggesting an increase over time in human stress at this site (note that with this result, there is only a 13% probability that the trend could be the result of chance (random) variation in HDI).

Table 10. Human Disturbance Index (HDI) and an example component metric within HDI—roads—at stream ecological integrity sentinel sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Assessment points are ecoregional in form and are the 95th percentile value from 38 reference sites in the park. No results for these indicators were greater than the assessment points. Values for HDI and roads increase as condition in a stream deteriorates and values above or equal to the assessment point are in a non-reference condition.

Variable	Site	Trend	Trend (p)	Assessment point	Start date	End date	Sample size	Median	Mean	Standard deviation	Minimum	Maximum
Human disturbance Index	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.8816	8/15/2008	8/20/2019	3	0.234	0.228	0.012	0.214	0.236
	North Fork Flathead—Border	−0.0029	0.765	0.8816	7/11/2008	8/22/2019	10	0.124	0.145	0.057	0.078	0.236
	Swiftcurrent Creek	0.0142	0.135	0.8816	10/18/2007	8/22/2018	11	0.146	0.185	0.069	0.095	0.306
	Reynolds Creek	−0.0036	0.413	0.8816	9/11/2008	8/21/2018	10	0.183	0.188	0.022	0.143	0.212
	Ole Creek	–[b]	–[b]	0.8816	8/27/2009	8/23/2018	9	0.000	0.004	0.010	0	0.029
	Middle Fork Flathead	–[a]	–[a]	0.8816	8/28/2018	8/19/2019	2	0.130	0.130	0.001	0.129	0.131
Road (component index in HDI)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.5	8/15/2008	8/20/2019	3	0.273	0.273	0.000	0.273	0.273
	North Fork Flathead—Border	0.0213	**0.091**[c]	0.5	7/11/2008	8/22/2019	10	0.136	0.155	0.114	0.030	0.333
	Swiftcurrent Creek	0.0108	**0.059**[c]	0.5	10/18/2007	8/22/2018	11	0.121	0.115	0.063	0	0.182
	Reynolds Creek	−0.0039	0.663	0.5	9/11/2008	8/21/2018	10	0.217	0.250	0.056	0.200	0.333
	Ole Creek	–[b]	–[b]	0.5	8/27/2009	8/23/2018	9	0.000	0.000	0.000	0	0
	Middle Fork Flathead	–[a]	–[a]	0.5	8/28/2018	8/19/2019	2	0.076	0.076	0.021	0.061	0.091

[a] The period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] No trend model was possible at Ole Creek given very low variability in the data.

[c] Significant trend ($p \leq 0.10$)

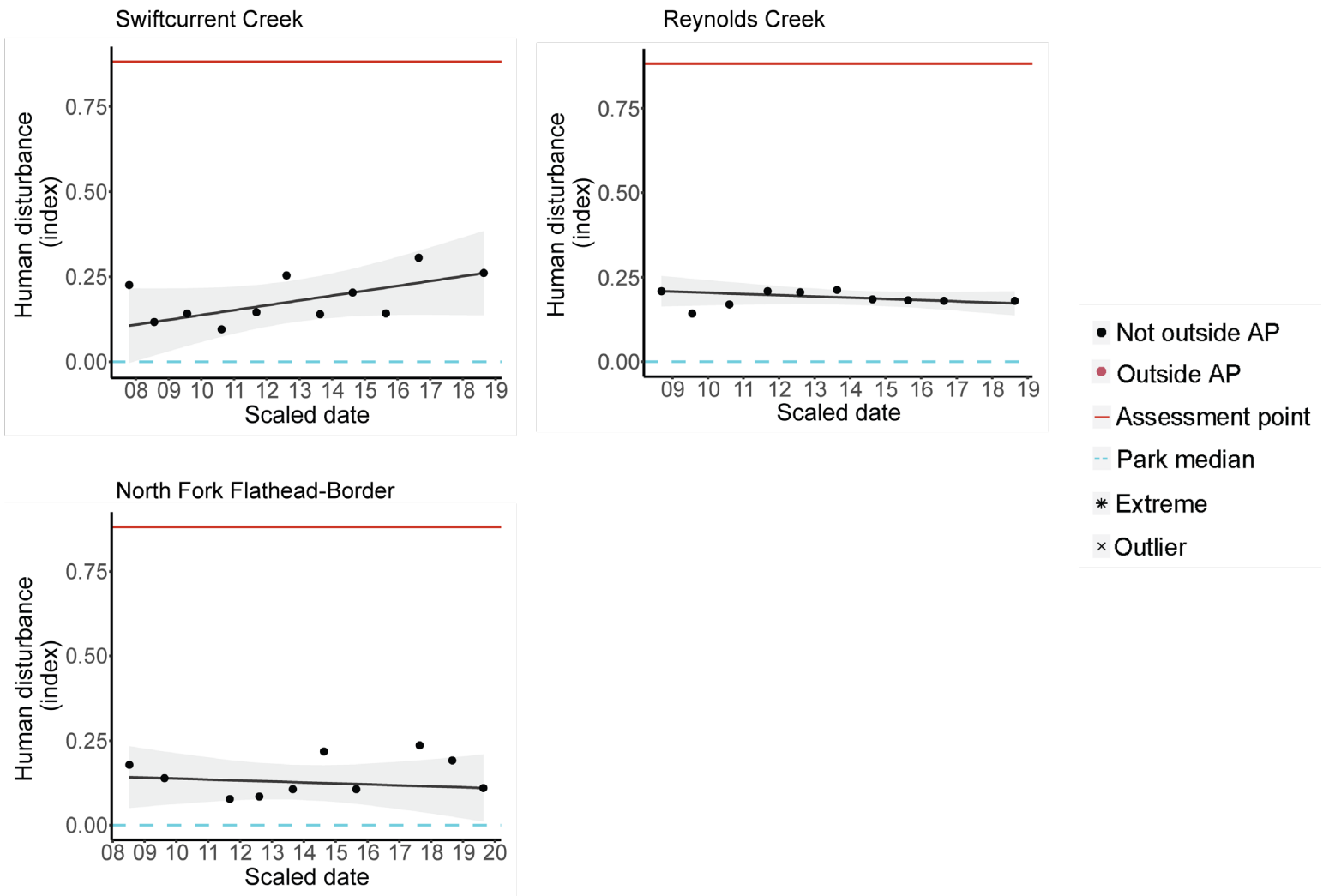


Figure 17. Human Disturbance Index (HDI) in Glacier National Park streams from 2007 to 2019. Swiftcurrent Creek (top left), Reynolds Creek (top right), North Fork Flathead—Border (bottom left). No model was possible at Ole Creek because HDI was zero or nearly so with all sample events. As available, red lines at the top of the figure give assessment points and dotted blue lines at the bottom of the figure give the park median. The fitted line (not statistically significant at any of these three sites) and its pointwise 95% confidence interval (light gray band) are from a quantile regression model at the median of the response vs. scaled date. Legend contains symbols that may not apply to this figure.

Across sentinel sites, HDI averaged 0.15. Ole Creek was almost always 0, with only occasional indicators (such as small pieces of trash) suggesting any human use. Reynolds Creek, just down slope from a large parking lot and a busy visitor facility, has on average the highest HDI, but Swiftcurrent Creek was close behind and had the highest maximum HDI value.

Patterns in the road component metric were similar to HDI, with generally low values, but only Ole Creek had no road disturbance in or near the site. The field-data based road metric did have two significant trends, increasing at North Fork Flathead—Border and Swiftcurrent Creek (Figure 18).

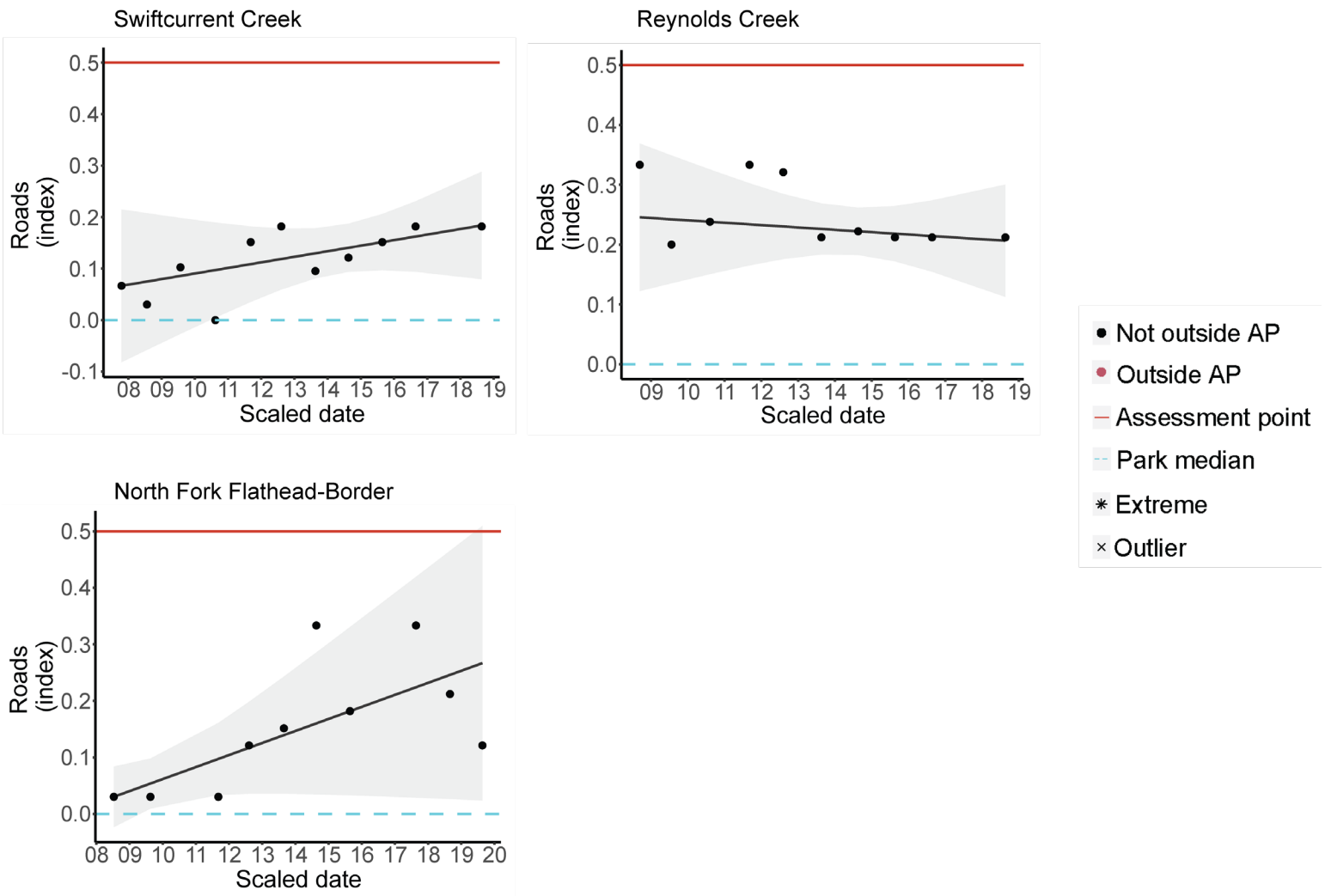


Figure 18. Roads as a component in the Human Disturbance Index in Glacier National Park streams from 2007 to 2019. Swiftcurrent Creek (top left), Reynolds Creek (top right), North Fork Flathead—Border (bottom left). No trend model was possible at Ole Creek because HDI was zero or nearly so with all sample events. As available, red lines give assessment points and dotted blue lines give the park median. The fitted line (not statistically significant at any of these three sites), and its pointwise 95% confidence interval (light gray band) are from a quantile regression model at the median of the response vs. scaled date. Legend contains symbols that may not apply to this figure.

Sentinel sites added in 2018 on North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression (QR) models as presented in this report for other sentinel sites. These sites did have higher scores for both HDI and roads, given their proximity to visitor facilities and developed areas outside the park (see Figure 8). Both rivers are used for recreation as well, and this is reflected in HDI scores.

3.2.6 Discussion and management application

Human land use in and around a stream can have a variety of effects. These effects include compacting and eroding soils, which can change water runoff patterns and bring nutrients and sediments into the stream, changing both its chemistry and clarity. Increased use in the backcountry can mean human waste and pathogens could likely enter streams and rivers. Land use changes in neighboring communities adjacent to parks can have increased nutrient and sediment loadings into park streams. These disturbances can contribute to changes in stream flora and fauna. For example, increased nutrients and sediments will change community composition of streams and can increase the presence of invasive species. Additionally, increased sediment levels can change fish habitat and make streams shallower, resulting in water temperatures rising at a faster rate. Understanding human disturbance components is critical to understanding stream status and trends.

In general, as with most sites in Glacier, there was minimal human disturbance at sentinel sites in 2007–2019. Moreover, all scores are relative to the disturbance gradient in the park, so even the higher values in the park are much lower than those for a stream in an urban corridor or a heavy agricultural area. The increasing trend in the road component metric at the North Fork Flathead—Border site is evident in a careful review of aerial imagery around the site, with small access roads to new homes and timber harvest areas appearing over time (Figure 19). At Swiftcurrent Creek there were no new roads constructed over the period of record that might explain the increasing trend. However, the metric is scored as the level of interaction of the site with roads, and field data show that as the channel shifted (i.e., in response to the beaver dam in place in 2014–2016), some riparian plots shifted closer to campground roads and thus were placed into a higher road disturbance level.

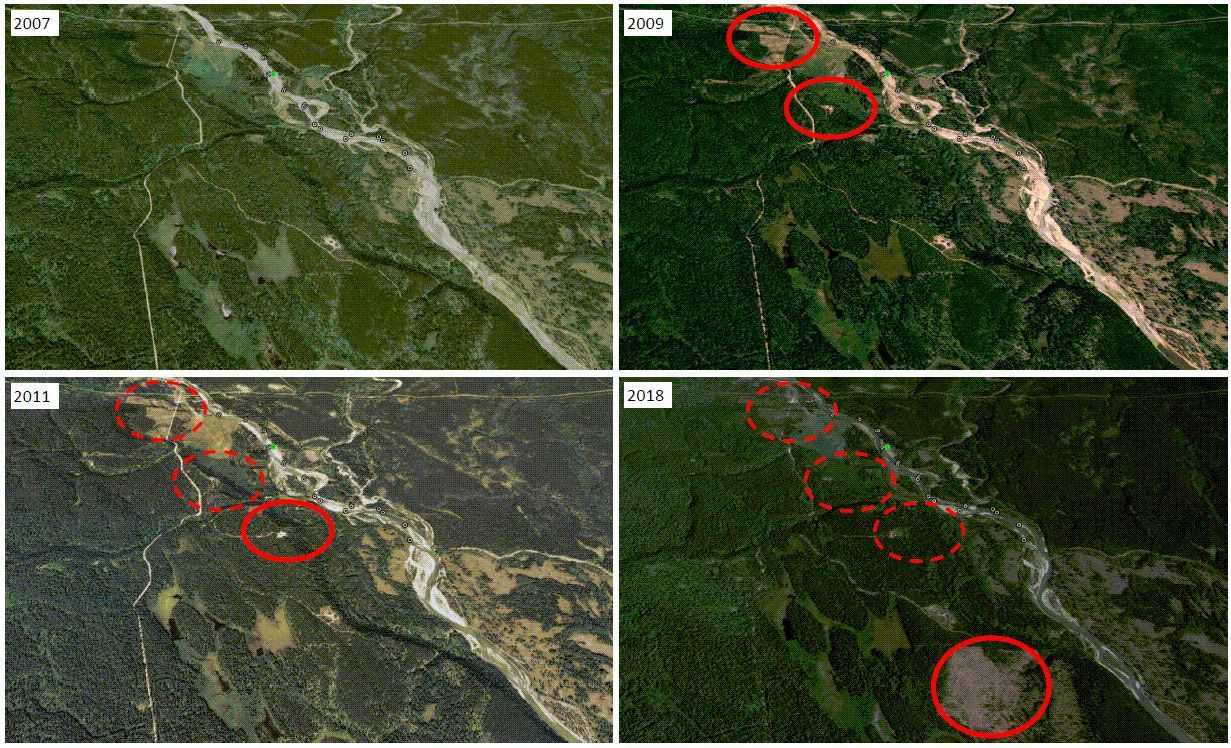


Figure 19. Road component metric of the Human Disturbance Index at North Fork Flathead—Border sentinel stream monitoring site in Glacier National Park. 2007 photo shows the start of Rocky Mountain Network period of record. 2009 photo shows a new inholding road and a road to small timber harvest (solid red circles, dashed circles show accumulation of new roads over the period of record. 2011 photo shows new inholding road. 2018 photo shows new roads to large timber harvest. The network sample reach is indicated by the series of gray circle points showing transect locations from near the top of the images to about halfway down the river channel.

There is a possibility of observer effects in the HDI. However, in the 2007–2009 baseline survey, HDI had a high signal to noise ratio (12.4; Schweiger et al. 2018). This suggests the variance in HDI is mostly across sites (or the signal we hope to estimate) versus within sites and across time (or noise). This suggests that HDI is less affected by random variation, such as from observer bias, and is more likely to show a consistent response to its drivers (Stoddard et al. 2008). We do not have specific QAQC data for sentinel site HDI data in 2007–2019 but expect observer impacts to be even lower in data that are measured every year (and are thus more familiar) as the data collection is often done by the same crew. Coupled with the corroboration in the imagery data for the North Fork Flathead—Border site that we show above, we feel the HDI and its components, like road proximity, are likely reflecting real pattern at our sentinel sites.

3.2.6.1 Park visitation

During the past decade, popularity of national parks has been increasing, with many parks observing record visitation. Glacier National Park is no exception to this phenomenon. There is a relationship between this increasing visitation and the disturbance a stream experiences as measured in the HDI, but this is complicated and the HDI should not be thought of or used as a direct proxy for sentinel

stream site visitation. Given the importance of visitor use per se, we introduce the subject briefly here and make connections to our stream monitoring as possible throughout this report.

Glacier National Park has a rich history of anthropogenic use dating back up to 12,000 years with Native American Tribes that inhabited the area. More recently, Glacier has been in the top ten most visited parks and has experienced a 60% increase in visitation during the past decade (NPS 2021). The majority of visitation occurs in the summer months between May and September each year, with the most popular month being July. Between 2008 and 2015 Glacier averaged 2.12 million visitors per year. This average has risen to 3.09 million visitors per year between 2016 and 2019 (Figure 20). While increased visitation meets one component of the NPS mission, it comes with challenges. Specifically, increased visitation can reduce the quality of visitor experiences, increase entrance station waits, crowd roadways and parking lots, and lead to more vehicle emissions and trampled vegetation and erosion, all of which have cascading impacts on park resources.

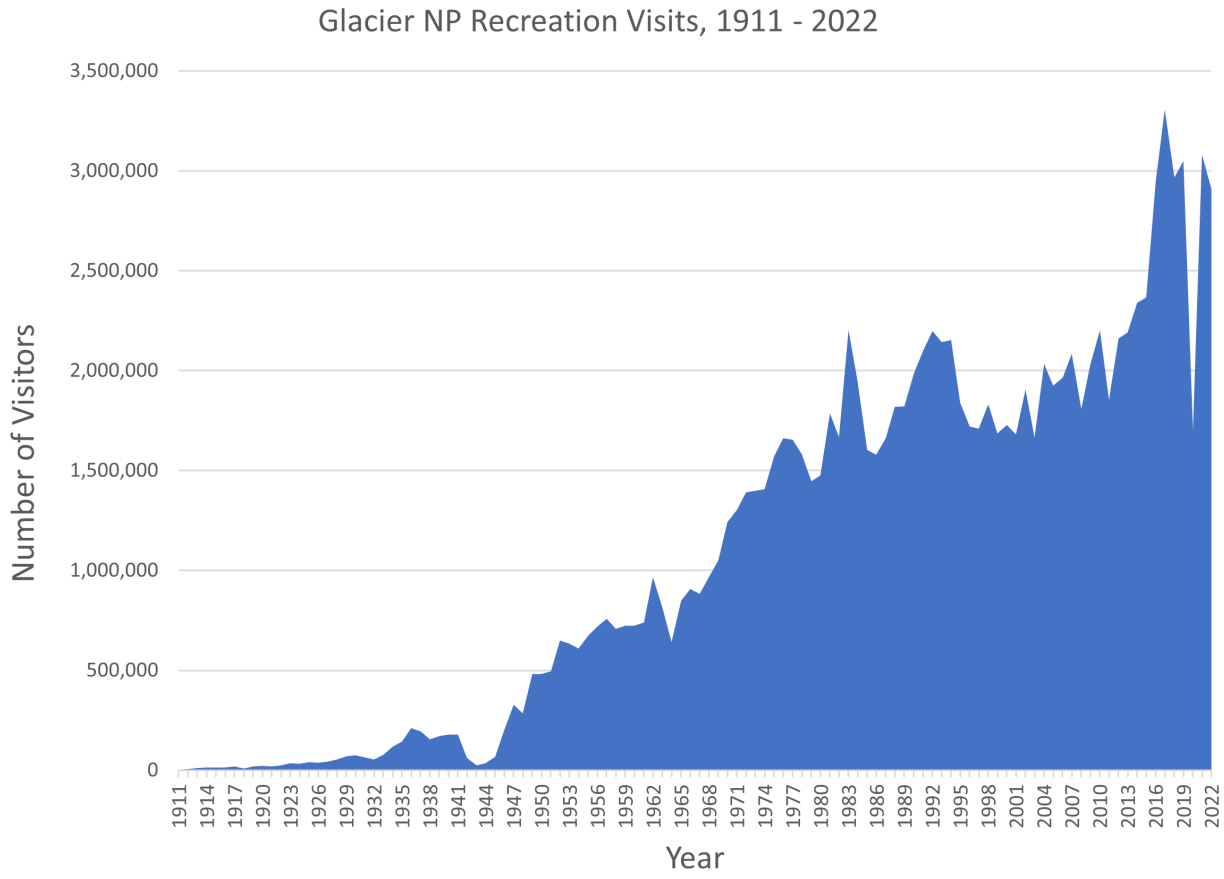


Figure 20. Annual Glacier National Park visitation, 1911 to 2022. Source: NPS Stats, National Park Service Visitor Use Statistics: STATS - Park Reports (nps.gov)

Land use changes outside of the park's jurisdiction are another source of impact. For example, active development around the park boundary can create new sediment inputs and change runoff patterns inside the park. The increase in park visitation likely reflects an increase in the popularity of the

region. This is evident at select areas just outside the park (see Figure 19) via increased road density and land use change.

3.3 Water and sediment chemistry

3.3.1 Introduction

The chemical environments of stream water and surficial bed sediments (hereafter "sediment") are the primary media for aquatic biota. The influence and role of chemistry in the condition of a stream is much like how the chemical composition and physical attributes of the atmosphere ("air") affect terrestrial organisms. Documenting the status and trends in chemistry is a fundamental part of monitoring streams and this has been done for decades within multiple long-term ecological monitoring programs (Figure 21).



Figure 21. Collecting water chemistry samples on Reynolds Creek, Glacier National Park, 2011. NPS / BILLY SCHWEIGER.

We have far more chemistry results than can be included in this report in detail. We therefore focus on results from a subset of parameters that we feel are most important or best illustrate patterns in trends at our sentinel sites. Results for additional parameters are available from the network on request.

QAQC is an important component of our stream monitoring in network parks. It has particular importance for chemistry data, with adjustments possible based on duplicates, blanks, and other QAQC lab analysis results. We present QAQC results for chemistry data in a future publication (in preparation by Schweiger and others).

3.3.2 Summary condition

Summary condition statements condense chemistry results into concise overviews of key aspects of trends, assessment, and possible management applications or concerns (Table 11).

Table 11. Summary condition table for water and sediment chemistry at Glacier National Park sentinel sites in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Overall water and sediment chemistry	17 indicators	The chemical environment of streams is the primary medium that aquatic biota inhabit and has direct and indirect effects on stream condition. Documenting the status and trends in water chemistry is a fundamental part of monitoring streams and this has been done for decades within multiple long-term ecological monitoring programs. Trends in water and sediment chemistry at Glacier National Park sentinel sites from 2007 to 2019 were largely stable, with most parameters changing little. Moreover, most parameters were within a reference range, meaning concentrations were not beyond assessment points established to identify potential problems and to classify status and trend at sentinel stream sites in the park. These results were expected for the Outstanding Resource Waters (ORW) of Glacier National Park. However, there are a few potential emerging issues. Most notably, water temperatures in park streams have increased and are expected to continue to increase given climate change projections. This parameter alone could push our overall water chemistry rating to "Caution," especially if the interactions between water temperature and hydrologic processes and water chemistry are confirmed with additional data and research. For example, soluble metals may increase to unexpected levels at our alpine sentinel site, likely driven by complex climate change processes, such as increased acidification at our sentinel sites. There are also some indications that aerial deposition of nitrates is interacting with climate-driven changes. We may also see slightly elevated major ion concentrations from visitor services, such as road maintenance, and these will likely have important interactions with climate processes. Many chemistry vital signs need additional data, analyses, and interpretation and our confidence in our overall rating is medium.	

Table 11 (continued). Summary condition table for water and sediment chemistry at Glacier National Park sentinel sites in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
In-situ water chemistry	Water temperature (continuous)	Water temperature is a critical driver of stream ecological condition, affecting the physiology of aquatic plants and animals. It is a key parameter monitored in Glacier National Park. Continuous temperature monitoring began at sentinel sites in 2009 and we present detailed analyses through 2019. There were significant increasing temperature trends at three of five sites that have long-term records associated with them. Warming water in the cold water streams in Glacier generally implies deteriorating environmental conditions for stenothermic cold water organisms typical of Glacier, which are intolerant of warmer water (thus the down arrow). Water temperature trends were similar across sites, suggesting meaningful generality in our estimates. Water balance and/or hydrologic drivers explained a large portion of the variation in maximum daily water temperature. Our models also suggest increases in water temperature may be accelerating; currently they are on the order of 1 to 1.5 °C a decade. In Glacier streams, many summer stream temperature daily maxima are above ecological management assessment points that are based on aquatic invertebrates and bull trout habitat suitability. Our data suggest several sites in the park that were once thermally suitable for bull trout are no longer viable for this species. Our data from Reynolds Creek suggests that this alpine stream has suboptimal water temperatures for high-elevation stoneflies, but this assessment point is less rigorous, and more work is needed here. Bioassessment of these streams supports our general conclusions. Temperature-based macroinvertebrate bioindicators increased at all sites except Ole Creek over our period of record. At the North Fork Flathead—Border site, this was a strongly significant increasing trend—one of the strongest in our entire dataset—showing that a macroinvertebrate assemblage indicative of warmer water developed over time at this site. These concordant biological patterns provide additional weight of evidence for the importance of water temperature in influencing biota at our sentinel streams and rivers and likely the park's waters overall. Finally, of all the responses in the SEI protocol, water temperature may have the most direct and clearest connection to climate change. Water in Glacier, as the name suggests, is influenced by the park's iconic glaciers. As glacial meltwater-induced flows are reduced by climate change, streams, especially in the alpine, will become shallower, with less flow, elevating water temperature and increasing the risk to important aquatic species. Park resource management might consider mitigation efforts to help streams and rivers respond to temperature changes. There may be limited options given the large-scale drivers of this; however, simple things like ensuring that riparian cover is retained (i.e., by minimizing loss to fire in riparian areas), and minimizing reductions to base flow, for example by carefully managing any water diversions in the park, can help. The park is of course already doing many of these things—and these should be continued. Our confidence in our water temperature data, and our assessment of them, is high.	

Table 11 (continued). Summary condition table for water and sediment chemistry at Glacier National Park sentinel sites in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

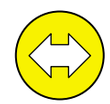
Vital sign	Indicator	Summary condition	Status, trend, and confidence
In-situ water chemistry (continued)	Dissolved oxygen	Sufficient dissolved oxygen (DO) is critical for biological processes in streams and rivers. High oxygen levels are especially critical for the metabolism of aquatic insects and salmonid eggs. There was no trend in DO at any sentinel site from 2007 to 2019. However, the warmer water we are seeing at sentinel sites reduces the amount of oxygen that can be dissolved and used in biological processes in a stream, and so ongoing monitoring of DO is warranted. We use a State of Montana minimum instantaneous DO concentration as an assessment point. DO levels *below* these assessment points are indicative of non-reference conditions. However, no DO data point was outside the assessment point at any site over our period of record. Our confidence in our DO data and in our assessment of patterns in these data are high.	
	pH	pH is a measure of the acidity in a stream or river and interacts with other water chemistry parameters. Because most aquatic species occur within specific pH ranges based on their physiology, changes in pH affect their physiological tolerances and thus may ultimately result in changes in species assemblages. pH determines the solubility and biological availability of many nutrients and heavy metals in water. For instance, most metals become more soluble at lower pH levels, and some of those become toxic to aquatic organisms. There was no trend in pH at any sentinel site from 2007 to 2019. The State of Montana establishes standards for pH in ORW waters such as those in Glacier National Park. Several pH values were outside (above or below) these assessment points during our period of record. These exceedances do not have any known direct cause from human uses and are more likely natural variability. However, Reynolds Creek had some episodic acidic conditions that may be related to increasing water temperatures and potentially atmospheric deposition of nitrates or sulfates. There is also some evidence of higher metal concentrations, perhaps related to this acidity (most metals become more soluble at lower pH values). The combination of pH and metals warrants some caution, ongoing monitoring, and more focused research. Given the complexity in assessing pH, especially its relationship with other responses, our confidence is medium.	
	Specific conductance	Specific conductance (SC) is a measure of total solute concentration in a stream or river. It is a useful indicator of how dilute or "clean" water is. Specific conductance had no trend at any sentinel site over our period of record, except at the North Fork Flathead—Border, where it continued an increasing trend seen in previous work. The trend at this site was significant and strong enough that we felt the overall trend warranted a deteriorating trend (down) arrow. There are no management, regulatory, or known ecological assessment points for specific conductance. Therefore, we estimate an ecoregional assessment point from reference sites that provide general guidance for this indicator under an expectation that Glacier streams should exhibit a reference state condition. Despite increasing specific conductance in the North Fork Flathead—Border, no water sample was outside the ecoregional assessment point at any site, suggesting that across our period of record, Glacier sentinel streams had SC below reference sites in the surrounding ecoregions. Work in landscapes with a more pervasive human footprint shows a clear relationship between elevated specific conductance and human disturbance. This may warrant some concern and ongoing monitoring. Our confidence in our SC data, and our assessment of them, is high.	

Table 11 (continued). Summary condition table for water and sediment chemistry at Glacier National Park sentinel sites in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

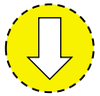
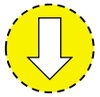
Vital sign	Indicator	Summary condition	Status, trend, and confidence
Major ions and sediment	Ions: chloride, sulfate, and calcium	The Rocky Mountain Network monitors several major ions at sentinel sites. Of these, chloride, sulfate, and calcium are important and useful indicators of increasing human disturbance. Regional long-term patterns in chloride show the positive impact of the Clean Air Act in a general sense (and the utility of long-term monitoring). Sulfate, on the other hand, has been increasing in mountain watersheds across the west and is interacting with changes in climate and hydrology. Most ion concentrations continued flat or negative trends from 2007 to 2019, as seen in previous work. The exception was at Swiftcurrent Creek, where both chloride and calcium had significant increasing trends. Recent evidence suggests a broad scale "salinization syndrome," with several ions (aka, salts) increasing in concentration, driven by climate change and human land use. This research suggests that recent changes in climate may be stimulating changes to hydrology and/or geomorphic processes, which in turn lead to accelerated weathering of bedrock and elevated levels of ions. While more research is needed in the park to support this, our results may suggest this is beginning to happen at select sentinel sites in Glacier National Park. There were also small but nonsignificant increases in sulfate at Reynolds Creek, Ole Creek, and Swiftcurrent Creek, which might also warrant caution if confirmed with additional data. There are no management, regulatory, or known ecological assessment points for these parameters. We estimate ecoregional assessment points from reference sites that provide general guidance for these indicators under an expectation of a reference state. No ions at any sites, except for sulfate at the North Fork Flathead—Glacier Rim, had any value above an ecoregional assessment point. However, we feel the increasing trends are potentially problematic, suggesting that localized impacts from land use and/or broad scale climate driven shifts may be becoming problematic. While this needs more data and research, it warrants some caution. The uncertainty in the assessment of these three parameters drops our confidence to low.	Chloride  Sulfate  Calcium 
	Alkalinity	Alkalinity is an important variable in a stream's ability to respond to acidification. It is also an indicator of relative system productivity. Evidence in several watersheds outside the park links changes in alkalinity to climate change. Alkalinity was generally stable at all sites, except for a marginally significant increasing trend at the North Fork Flathead—Border site. There are no management, regulatory, or known ecological assessment points for alkalinity. Alkalinity concentration may either be too low or too high—reference values are generally in the middle of a typical range in alkalinity. We estimate an upper ecoregional assessment point from reference sites and use a lower value from the US EPA that together provide general guidance for this indicator under an expectation of a reference state. The North Fork Flathead—Border site had multiple alkalinity concentrations above the upper assessment point. While large rivers can have naturally higher values, this may suggest some land use activities in the Flathead River watershed in Canada might be impacting the site. Alkalinity was occasionally below the lower assessment point at the Reynolds Creek site. Alpine watersheds are naturally poorly buffered, but coupled with data on pH and metal concentrations, this may also be of some concern. Given the complexity in assessing alkalinity, especially its relationship with other responses, our confidence is medium.	

Table 11 (continued). Summary condition table for water and sediment chemistry at Glacier National Park sentinel sites in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Major ions and sediment (continued)	Total suspended solids	Total suspended sediment (TSS) is a measure of the amount of particulate matter in a stream or river. Sediment can have a direct impact on stream biota and can be associated with elevated levels of nutrients and metals. Most TSS in the park's streams and rivers is from natural sources. Trends in TSS over the period of record were mixed. Two were increasing and two were decreasing, but none were significant, and levels are relatively low in these systems to begin with. There are no management, regulatory, or known ecological assessment points for TSS. We estimate an ecoregional assessment point from reference sites that provide general guidance for this indicator under an expectation of a reference state. Glacier sentinel site streams over our period of record had low TSS and were as sediment free as the best sites in the surrounding ecoregions. The North Fork Flathead—Border occasionally had some excess TSS. This is most likely from natural sources, as it was driven by spring runoff and should be watched. TSS is challenging to collect and analyze, and this drops our confidence to low.	
Nutrients	Total nitrogen and phosphorous	We measured multiple nutrients (including nitrogen (N) and phosphorous (P)) at our sites across the park. Nutrients are essential for stream ecological function. Total N and P have been shown for decades to limit productivity in streams. However, elevating concentrations cause eutrophication, which can cause a variety of problems with primary productivity. This can degrade ecosystem integrity and hamper recreational activities, as well as impact human health. Trend coefficients for nutrients were mixed, with some sites showing small increases and other sites with small decreases and no model suggesting a significant trend in total nitrogen or total phosphorous. Montana DEQ has developed a set of regulatory nutrient criteria for total nitrogen and phosphorous that are specific to summer months and ecoregion. We use these as assessment points. There was a single exceedance of the assessment point for total nitrogen at Swiftcurrent Creek in the spring of 2010. This is likely due to naturally elevated levels of nitrogen during runoff. Concentrations of total phosphorous at the North Fork Flathead—Border were above the assessment point in 15% of our results. Nearly all of these also occurred during spring runoff when nutrient concatenations can be naturally elevated. However, this result may warrant some caution should additional data confirm and extend this result to more seasons. Across the Rocky Mountain region over the last decade higher levels of phosphorous have been measured in remote wilderness watersheds. Remote watersheds may be disproportionately influenced by elevated phosphorus from climate change driven increases in runoff. As much of Glacier is remote, patterns in nutrient levels warrants more research. A nutrient specific bioindicator for macroinvertebrates (the Hilsenhoff Biotic Index; HBI) was often above its assessment points, especially at the Swiftcurrent Creek and North Fork Flathead—Border sites. This biological response may suggest an effect of elevated nutrients on stream condition during select sample events at these sites. However, overall, HBI decreased across our period of record at the North Fork Flathead—Border and had no trend at Swiftcurrent Creek. Application of the state's integrated nutrient protocol (see Integrated Nutrient Assessment summary below) indicates there is no nutrient impairment at our sites. Given the complexity in assessing total phosphorous, our confidence for this nutrient is medium. We have higher confidence in our data and assessment of total nitrogen.	Total N  Total P 

Table 11 (continued). Summary condition table for water and sediment chemistry at Glacier National Park sentinel sites in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Nutrients	Integrated nutrient assessment	Montana DEQ provides a detailed protocol for comprehensive evaluation of possible nutrient issues. This protocol uses a tiered approach based on nutrient concentrations, algae-based productivity, a diatom nutrient increaser metric, and a macroinvertebrate nutrient indicator metric. The Rocky Mountain Network SEI protocol provides data suitable for use in the Montana DEQ method, but, given smaller than specified samples sizes, the ability to infer nutrient impairments is limited. As an explicit regulatory objective is not part of our monitoring protocol, a smaller sample size may be acceptable. Should a network sentinel site ever appear at risk of a serious nutrient exceedance, with a clear connection to a human cause, we can adjust sampling intensity accordingly. There were no significant or marginally significant trends in the integrated nutrient assessment at any sentinel site over our period of record. No years at any sentinel site had concentrations of nutrients significantly above Montana DEQ criteria. Likewise, no sites had any exceedances in chlorophyll-*a*/ash free dry mass samples, or the diatom increaser metric. However, several sites had macroinvertebrate nutrient indicator metric values above the criteria. Where we occasionally see elevated total phosphorous levels coupled with high values of the macroinvertebrate nutrient metric but not the diatom increaser or productivity metrics, its possible sampling timing may have missed algal peak, and/or there may be possible false negative or results close to the decision threshold. This may describe some years at the North Fork Flathead—Border site and suggests ongoing monitoring and evaluation of all the factors involved in nutrient assessment is warranted. Given the complexity in some competing results within the integrated assessment, our confidence is medium.	

Table 11 (continued). Summary condition table for water and sediment chemistry at Glacier National Park sentinel sites in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/ warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Trace elements: metals in water and sediment	Arsenic, cadmium, copper, zinc, and mercury in sediment	Metals occur naturally in most soils and bedrock in and around the park. Movement of metals into streams from these sources can occur from natural rock weathering and forest fires. There are several human sources of metals in more human influenced landscapes. However, unless mining begins in the Canadian North Fork in the future, this is and will continue to be absent in the park. The direction and significance of trends in sediment metals were mixed. However, of the 15 models in this group, five were significant, four with increasing concentrations and thus declining condition. Specific results include significantly increasing copper and zinc at Swiftcurrent Creek, cadmium at Reynolds Creek, and arsenic at North Fork Flathead—Border. In contrast, arsenic at Ole Creek significantly decreased. There are no published regulatory standards for metals in sediments; however, consensus based expert opinion criteria have been developed or another ROMN park, Grant Kohrs Ranch National Historic Site. We informally apply these to Glacier but caution that because they were developed for systems where metal contamination is expected, they are likely conservative or define thresholds at a higher concentration than where true metal-based issues might occur in a wilderness park like Glacier. If these assessment points are exceeded in a landscape like Glacier, it would likely be a meaningful result. Arsenic was above the assessment point at three out of six sites, with the percent above the assessment point ranging up to 55.6% at Reynolds Creek. This may reflect natural background levels based on the geology of these drainages. However, arsenic levels at Reynolds Creek are above an assessment point that is in all likelihood conservative for an alpine stream in the middle of a wilderness park with no identifiable anthropogenic sources. Moreover, higher levels of arsenic were generally consistent across all sentinel sites from 2007 to 2019. This is a surprising result. More focused work should examine the potential sources of and biological responses to these metals in Reynolds Creek and other sentinel sites in Glacier as climate-induced degradation of water quality in watersheds draining mineralized areas has significant implications for the ecological integrity of these systems. Given our use of an assessment point designed for other systems and the more limited sample size of metals in sediment, our confidence is medium.	Cd, Cu, Zn, Hg  Arsenic 

3.3.3 In situ physiochemistry: continuous water temperature

The following sections present results and interpretation of models of continuous water temperature in relation to climate, hydrologic, and water balance predictors. Given the importance of water temperature and the unique continuous data we collect in the park to monitor it, there are more details in this section than others in this report. However, the summary condition statements, presented at the beginning of Section 3.3, still include key take home messages for our water temperature monitoring and may be used if the details in the following are of less interest. Appendix B summarizes the development of the data used in our water temperature models.

3.3.3.1 Background and relevancy

Water temperature plays a key role in the ecology of streams (i.e., Kishi et al. 2005; Isaak et al. 2016; Giersch et al. 2016; Jordan et al. 2016). It controls or is related to multiple processes (Isaak, Luce Horan et al. 2018), from primary and secondary productivity, ecosystem-level trophic state, assemblage composition, individual organism behavior and physiological response, and important physiochemical interactions, such as the toxicity of chemical compounds. Shifts in stream temperature can act as a stressor on aquatic species, directly affecting their physiology, and secondarily, by impacting stream processes that most organisms rely on for their survival (e.g., Jones et al. 2017; Poole and Berman 2001; Schülting et al. 2016).

Water temperature responds to climate drivers—air temperature, precipitation, and the interaction of these, as manifest in water balance (water cycle). Temperature in streams and rivers is also influenced by multiple proximate factors including discharge, watershed size and/or topology, land use, and groundwater input (Sanders et al. 2017). Local-scale variables, such as shade and macrohabitat complexity (i.e., beaver created features, large wood, etc.) also influence, at least in part, water temperatures. Interactions in these drivers across spatial scales, from regional to local and across time (i.e., lags by season or year), ultimately determine patterns in water temperature in a stream.

Rocky Mountain Network long-term monitoring of water temperature at Glacier National Park sentinel sites supports park management of water temperature under changing climate regimes (Briske et al. 2015). This is accomplished by providing park management with estimates of how water temperatures are changing over time and elucidating the important drivers of these patterns. The role of water temperature has important implications in the park, especially given the ongoing loss of glacier, meltwater-fed systems (Rood et al. 2005) that cool many of the park's streams and rivers.

3.3.3.2 Assessment points

Surprisingly, there are no explicit regulatory water temperature standards from Montana DEQ or US EPA for water temperature. Narrative standards as part of the state's Nondegradation Policy (75-5-303, Montana Code Annotated 2021) for Outstanding Resource Waters state that "existing uses of state waters and the level of water quality necessary to protect those uses must be maintained and protected" and this can be applied to water temperature. However, as a narrative standard, it is difficult to interpret actual data. It is also focused on point-source discharges that do not exist in the park. Temperature criteria do exist in neighboring states. For example, the State of Idaho (IDEQ

2016) has established temperature criteria for three aquatic life use categories based on fisheries data. However, these are likely only marginally relevant to Glacier, and we do not use them here.

We therefore develop two aquatic species-focused ecological thresholds as assessment points. Both *Lednia tumana* and *Zapada glacier* have been listed under the US Endangered Species Act (USFWS 2021). Muhlfeld et al. (2011) and Jordan et al. (2016) showed compelling evidence that climate change links changes in the fate of effects on endemic macroinvertebrates like the meltwater stonefly, *Lednia tumana,* to the fate of glaciers in Glacier National Park as a result of shifts in water temperature extremes and phenology. Under a future (yr. 2030) climate scenario, the projected distribution of *L. tumana* was shown to dramatically contract, with predicted loss of over 80% of its current range. Several other species in alpine streams in the park face similar problems (Giersch et al. 2015). We extract assessment points for *L. tumana* based on its predicted absence from sites with mean and maximum temperatures exceeding 10 °C and 18 °C, respectively. We recognize that the thermal tolerances of this and other alpine species are more complex (Hotaling et al. 2017) than these single numbers imply, but this is a start. We apply these assessment points to our Reynolds Creek site as *L. tumana* have been recorded at this site (J. Giersch, USGS biologist, pers. comm. 9/2019), but they are uncommon and/or do not appear in our data as species in this genus are difficult to identify without adults. These assessment points are similar to tolerances developed for a suite of European alpine macroinvertebrate taxa, many of which are cosmopolitan and occur in Glacier National Park (Niedrist and Fureder 2021).

For sites where *Lednia tumana* is not expected to occur, we use an assessment point for bull trout, *Salvelinus confluentus,* based on Selong et al. (2001) and Rieman and Chandler (1999). This research documented peak feeding and growth temperatures in the range of 10.9 to 15.4 °C. More specific to the Glacier National Park ecosystem, Jones et al. (2017) developed an optimal thermal range for feeding and growth during the month of August (when stream temperatures are at their maximum) between 10 and 14 °C. Ole Creek and the Flathead River system are likely historical bull trout fisheries (USFWS 1998; C. Downs, NPS biologist, pers. comm. 10/2022) and we apply these minima and maxima values to August maximum daily temperatures at these sites. Swiftcurrent Creek presents a bit of a challenge as technically it is not a historical bull trout fishery given a large waterfall barrier just downstream from our site. However, following advice from Glacier National Park staff (C. Downs, NPS biologist, pers. comm. 10/2022) we also use the bull trout assessment point as well. This assessment point is likely protective in general of the stream, and without the fish barrier, the stream would likely have been a bull trout fishery. Swiftcurrent Creek is also representative of other streams on the park's east side that have bull trout populations where similar dynamics may be at play. The application of the assessment point to Swiftcurrent Creek might serve as a qualitative proxy for these other streams in the park.

3.3.3.3 General patterns across sites

Water temperatures at our six sentinel sites were typical of high elevation flowing water in the Northern Rockies, with cold water at all times of year except ice-free summer days (Figure 22). Temperature varied by day of year in a strongly non-linear but intuitive way, increasing through spring into summer then falling off in the fall. Winter water temperatures (not used in this report)

were consistently near zero. All sites were perennial, with year-round water flow preventing temperatures from dropping meaningfully below 0 °C, even during the winter.

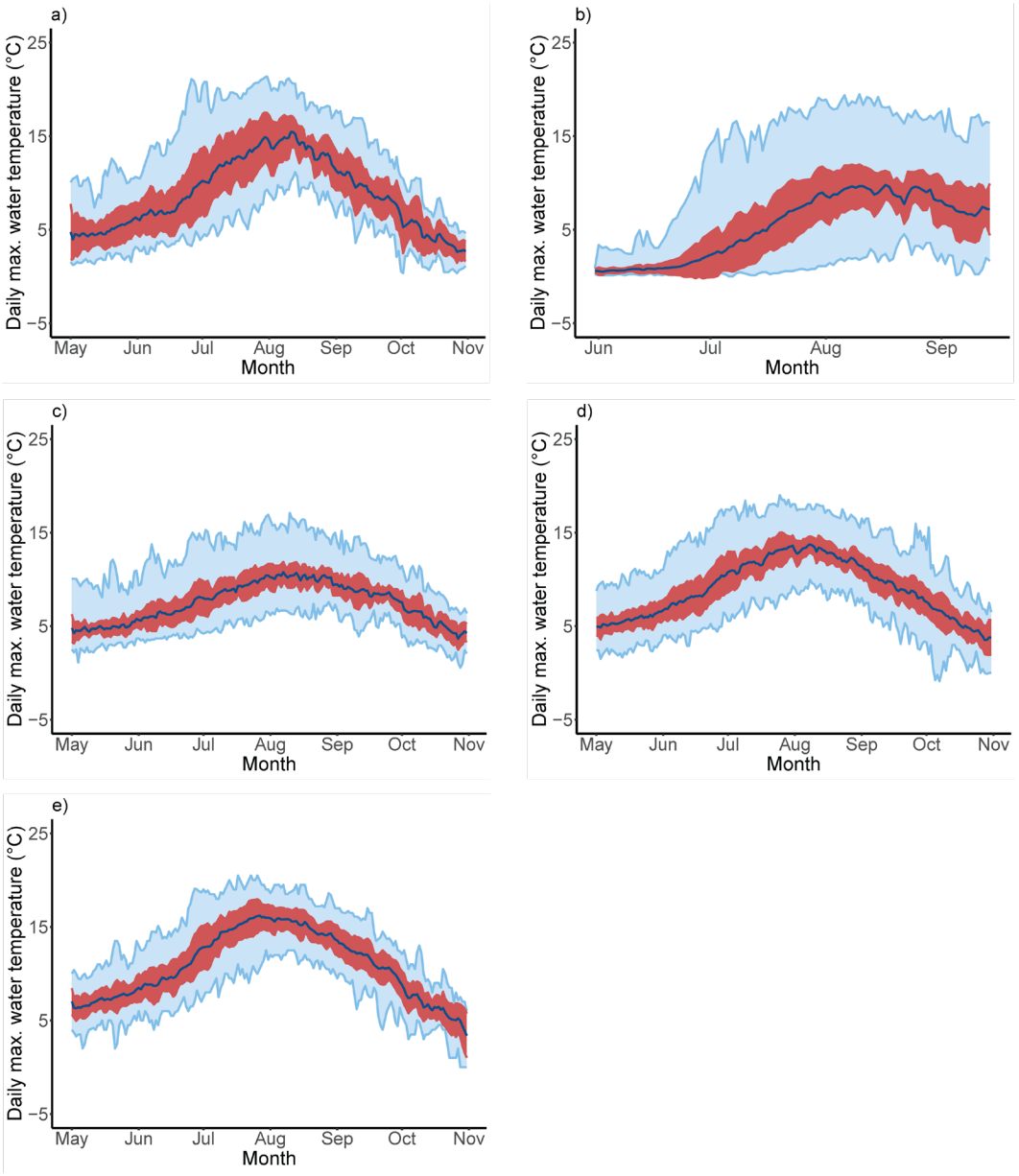


Figure 22. Mean daily maximum stream temperature during ice-free seasons at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. Swiftcurrent Creek (a), Reynolds Creek (b), Ole Creek (c), North Fork Flathead River—Border (d), North Fork Flathead—Glacier Rim (e). We do not show the Middle Fork Flathead site, given a shorter period of record; it will be similar to the North Fork Flathead—Glacier Rim site. Solid black line shows the daily mean maximum water temperature across years in each period of record bounded by its minima and maxima (Min/Max, outer blue shaded region) and its standard deviation (standard deviation (SD), inner red shaded region). Each time series is trimmed to an approximate ice-free period specific to the site.

Water at the Reynolds Creek site was considerably colder year round and during ice-free periods than at our other sites. Reynolds Creek is in the alpine of Glacier National Park and thus has colder air temperatures and a longer ice- or snow-covered period. However, the site lacks canopy, is shallow, and has a small surface and groundwater catchment with less capacity to buffer temperature fluctuations. Thus, temperatures were often highly variable, with warm summer days (see the much broader bands for minimum/maximum and standard deviation in Figure 22). Our site at the North Fork Flathead—Glacier Rim near the southern edge of the park was considerably warmer than other sites. It is not clear why this location would be warmer than other sites. The river channel is large at this location and the loggers were in a littoral zone that may receive more incidental sunlight warming the water, but this needs more work to verify. We do not include the Middle Fork Flathead site, given a shorter period of record; however, we expect it will be similar to the North Fork Flathead—Glacier Rim site.

3.3.3.4 Trend

Models

We used General Additive Models (GAM) to estimate trend in maximum daily water temperature and to identify and describe predictors that might help interpret this important ecological response. Methods used in these trend models are presented in detail in Appendix C.

Temporal predictors in the models included a day of year (DOY) term, factorial season, and time scaled and normalized to range between 0 and 1 (hence the name "Scaled Date"). Because we did not model water temperatures across sites, spatially variable predictors like elevation or local canopy cover are not used. Site scale predictors often explain large amounts of variation across space in water temperature (Jones et al. 2014, 2017; Isaak, Wenger, Peterson et al. 2017; Isaak et al. 2020) and we will explore these model forms in the future.

Following Grace and Irvine (2020), we limited predictors to those that likely had a connection to water temperature (Briske et al. 2015; Sanders et al. 2017; Isaak, Luce, Chandler et al. 2018; Isaak, Luce, Horan et al. 2018). We included independent variables under hypotheses of a variable being either (or both) 1) drivers of water temperature, such as drainage or discharge (Isaak, Wenger, Peterson et al. 2017; Isaak et al. 2020), or 2) responses to water temperature, such as evapotranspiration or soil moisture (Kaandorp et al. 2019). Our intent was to best describe temperature time series and estimate how trend and both structural drivers and responses to temperature dynamics contribute to this.

Results

Trend results are presented in detail in Appendix C. Maximum daily water temperature increased at four out of five sites and these trends were significant or nearly so. *There was a remarkable increase of about a 1 to 1.5* °C *rise in temperature per decade at the four sites with meaningful trends.* From one to three climate or water balance terms appeared in each model, and these were consistently some combination of growing degree days (GDD), drainage or discharge, soil moisture, and actual evapotranspiration. Drainage was in three of the five models, somewhat surprisingly in place of continuous stream flow at Swiftcurrent Creek, where we had both these candidate variables. The watersheds for Ole, Swiftcurrent, and especially Reynolds Creeks are relatively small compared to

the two big river sites where continuous flow outperformed drainage in modeling maximum water temperature. Drainage and stream flow are strongly related, and both are hydrologic drivers; however, given the drainage term's calculation from water balance-based runoff, it may include more of a groundwater component, especially in the smaller watershed and gaining (i.e., with meaningful ground water input) stream reaches typical of our three smaller stream sites. Groundwater can have a strong impact on water temperatures (Kaandorp et al. 2019), and the improvements in the three smaller watershed models from including drainage may reflect this. Drainage may also be better connected to water temperature in non-baseflow seasons because of its water balance basis.

Overall, there was a high degree of similarity across sites in trend and the types of covariates included, suggesting meaningful generality in our estimates of trend in water temperature. All models explained a large portion of the variation in maximum daily water temperature at each site and included interpretable water balance and/or hydrologic drivers. The relative importance of date was generally low, suggesting that, while time was important in water temperature dynamics at many of our sites, trend across time explained less of the overall variation than other time related terms (season and day of year) or the water balance/hydrologic predictors in the models. All models included a strongly significant non-linear smoothed term for day of year (DOY), with a characteristic bell-shaped fit. This was likely the most important impact of using GAM models, as otherwise this term would have been difficult to fit. Season (as a factor or categorical term) was also in most models. Patterns across season were intuitive, with highest temperatures in the summer, although these differences were rarely significant. Partial residual plots (Figure 22) show trends at each site when all other terms are controlled for. These plots are the best way to visually interpret complex multivariate models such as GAMs.

3.3.3.5 Condition

Water temperature is a fundamentally important component of stream condition in Glacier. We apply bull trout and macroinvertebrate based assessment points to assist in the interpretation of status and trend. Stream water temperature is also part of our bioassessment using Glacier's macroinvertebrate and diatom communities (presented in detail below). In brief, bioassessment models suggest that there are changes in these communities at several sites likely in response to increasing water temperatures.

Daily maximum August temperatures were above the bull trout assessment point of 14 °C at Swiftcurrent Creek (83% of August days), Ole Creek (47% of August days), North Fork Flathead—Border (76% of August days), and North Fork Flathead—Glacier Rim (92% of August days) (Figures 24 to 28). Only Ole Creek had relatively few August days across our period of record unsupportive of bull trout populations (in terms of water temperature). Our trend models suggest that temperatures are increasing at three of these four sites, albeit in a clearly significant way only at Swiftcurrent Creek. Given barriers preventing movement of bull trout into our sample site, the park considers Swiftcurrent Creek unavailable to bull trout. We therefore treat Swiftcurrent Creek as more of an indicator for potential thermal conditions at similar streams. The bull trout water temperature assessment point is also likely protective of general conditions in streams like Swiftcurrent Creek, and this has general applicability. Species like bull trout have many adaptations to deal with variable

habitat suitability, and stream reaches have many microhabitats (i.e., tributary inlets, deep pools) where temperatures are likely colder (Lawrence et al. 2014). Bull trout may thus persist in streams that are often too hot. However, temperature also interacts with many other stressors, such as invasive species, loss of riparian cover, and excess nutrient concentrations, and these negative synergistic effects can overwhelm a species' ability to behaviorally compensate for high temperatures (Bell et al. 2021). Additional research and monitoring will be needed to determine if and how bull trout manage increased water temperatures at our sentinel sites, but the elevated thermal conditions are not likely beneficial to the iconic aquatic species of Glacier, like bull trout. Our trend models suggest that temperatures are increasing at Reynolds Creek, albeit slowly and in a nonsignificant way, suggesting this has and may continue to only get worse. Reynolds Creek, like most alpine streams, has little to no shading and therefore little localized protection from increasing temperatures. Models suggests Reynolds Creek temperatures are influenced most by hydrology and thus have likely strong connections to climate change. As we discuss below, climate is driving water temperature upwards at an alarming rate at most of our sites, suggesting a challenging future for *Lednia tumana* and many other alpine aquatic macroinvertebrates.

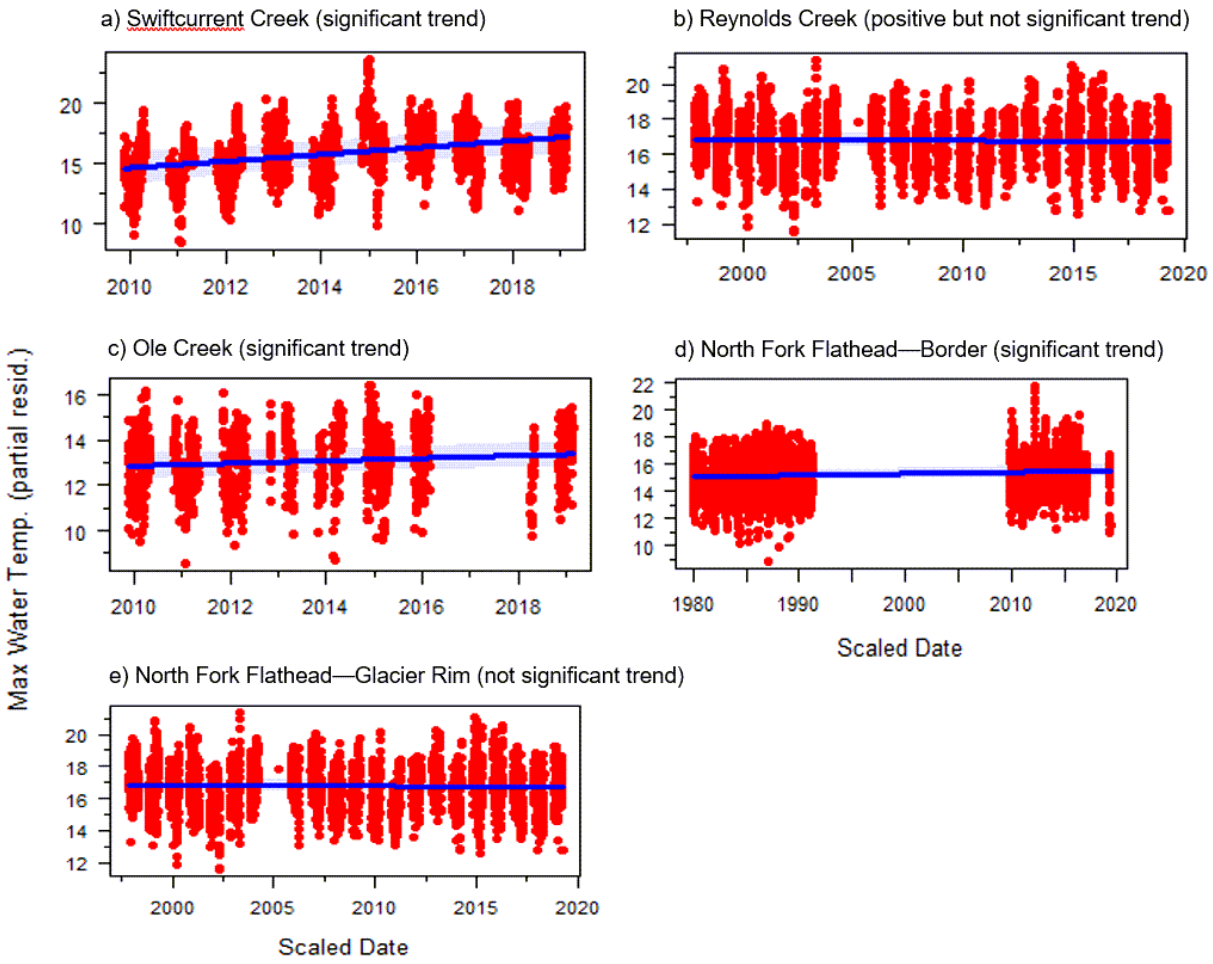


Figure 23. Sentinel site water temperature trend results, Glacier National Park. Swiftcurrent Creek (a), Reynolds Creek (b), Ole Creek (c), North Fork Flathead—Border (d), North Fork Flathead—Glacier Rim (e). Each panel shows trend as partial residuals of maximum daily water temperature that account for other independent variables in each model (see Appendix C). Gray shaded areas show the 95% confidence interval around each fit. Note different X and Y axis scales in each panel.

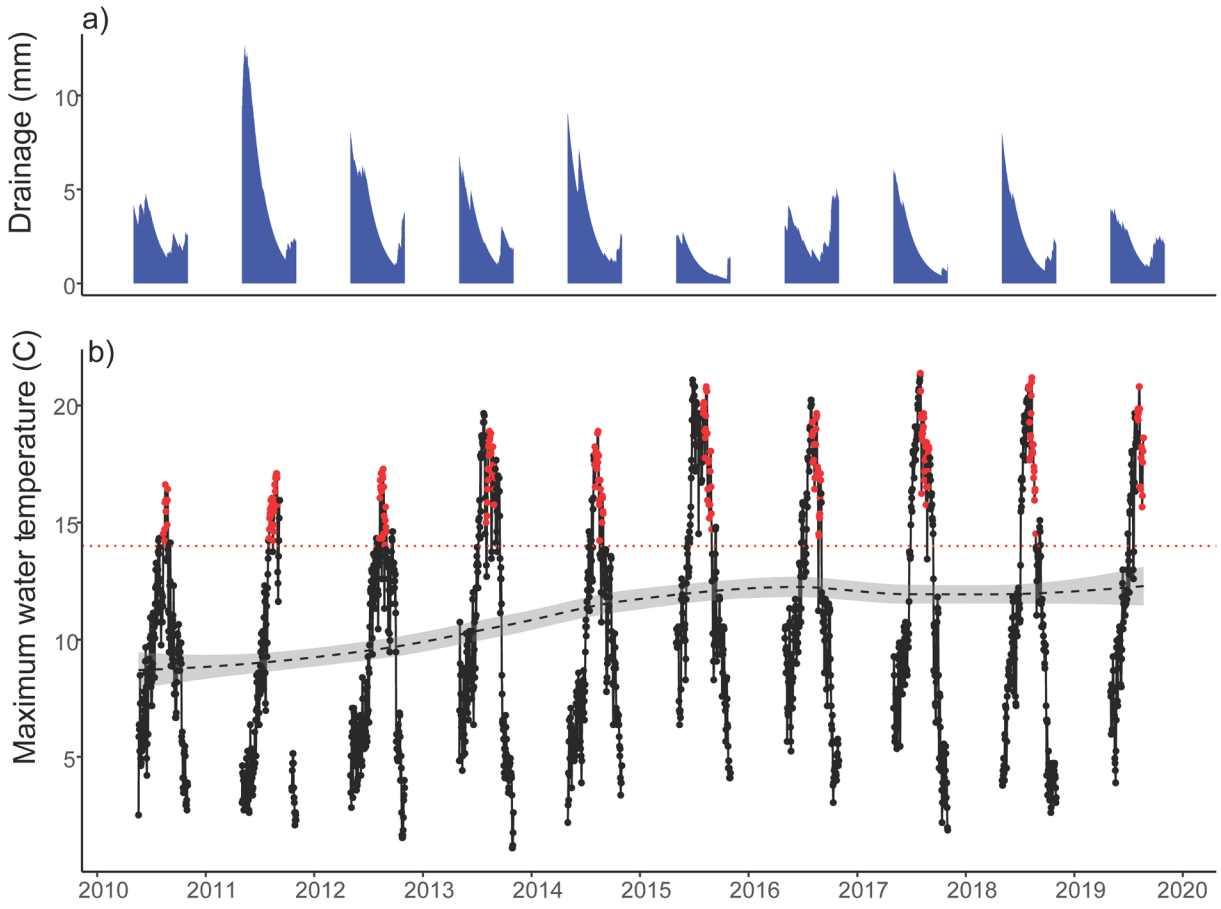


Figure 24. Daily stream temperature compared to an assessment point at Swiftcurrent Creek in Glacier National Park. The upper figure (a) gives drainage from a water balance model. The lower panel (b) shows daily maximum temperatures with a loess fit (black dashed line) and its 95% confidence interval (gray band). Red dotted line is a temperature assessment point for bull trout (*Salvelinus confluentus*) of 14 °C (Jones et al. 2014). August temperatures above this threshold are colored red. Note that the application of this assessment point to Swiftcurrent Creek is for illustration purposes only as the site is not a historical bull trout fishery where this assessment point is applicable.

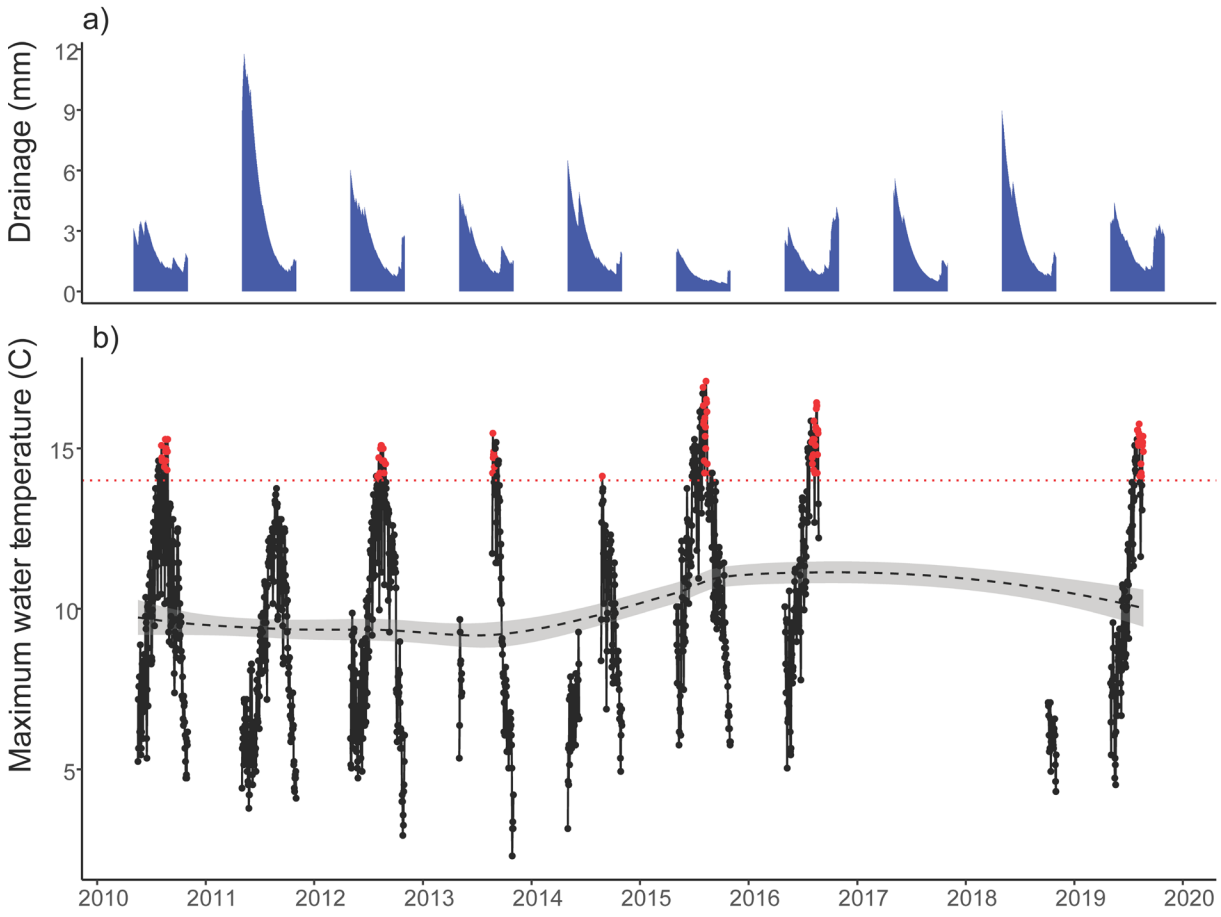


Figure 25. Daily stream temperature compared to an assessment point at Ole Creek in Glacier National Park. The upper figure (a) gives drainage from a water balance model. The lower panel (b) shows daily maximum temperatures with a loess fit (black dashed line) and its 95% confidence interval (gray band). Red dashed line is a temperature assessment point for bull trout (*Salvelinus confluentus*) of 14 °C (Jones et al. 2014). August temperatures above this threshold are colored red.

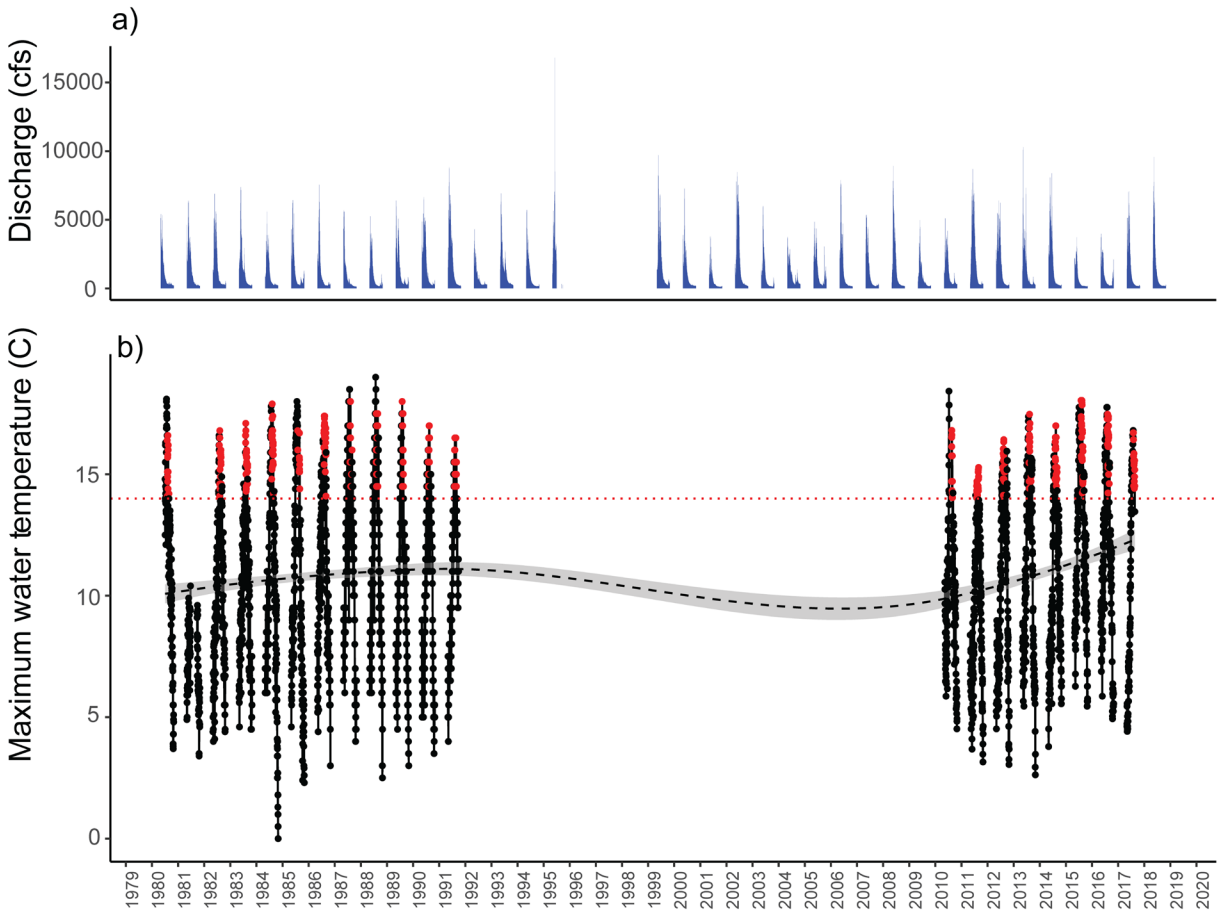


Figure 26. Daily stream temperature compared to an assessment point at the North Fork Flathead—Border site in Glacier National Park. The upper figure gives discharge from the USGS gauge at the site. The lower panel shows daily maximum temperatures with a loess fit (black dashed line) and its 95% confidence interval (gray band). Red dotted line is a temperature assessment point for bull trout (*Salvelinus confluentus*) of 14 °C (Jones et al. 2014). August temperatures above this threshold are colored red.

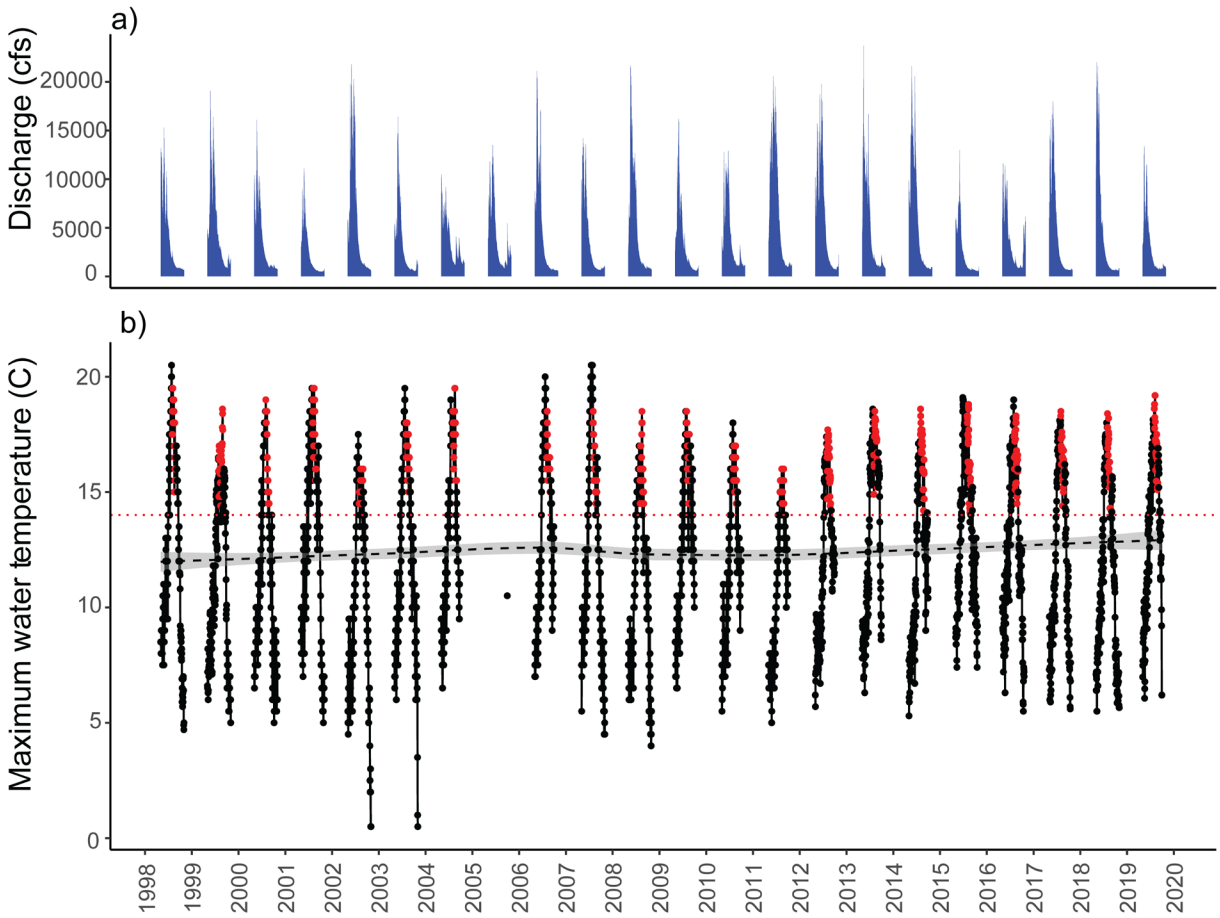


Figure 27. Daily stream temperature compared to an assessment point at the North Fork Flathead—Glacier Rim, site in Glacier National Park. The upper figure (a) gives discharge from the USGS gauge at the site. The lower panel (b) shows daily maximum temperatures with a loess fit (black dashed line) and its 95% confidence interval (gray band). Red dotted line is a temperature assessment point for bull trout (*Salvelinus confluentus*) of 14 °C (Jones et al. 2014). August temperatures above this threshold are colored red.

Daily maximum summer temperatures between 2009 and 2019 in Reynolds Creek (Figure 28) were above the assessment point for the federal threatened stonefly, *Lednia tumana,* 2.5% of the time. Based on a mean temperature assessment point, 15% of summer days were above the threshold. Muhlfeld et al. (2011) do not identify a more specific period where these thresholds are relevant or offer nuanced discussion of the differences between the mean and maximum assessment point. Treanor et al. (2013) suggest that *L. tumana* has a greater tolerance to rapid increases in temperature, perhaps as an adaption to the highly variable alpine thermal environment. This may suggest that our mean assessment point better approximates chronic thermal conditions of greater importance to *L. tumana*. If we restrict the period to August, as done with the bull trout assessment point, the percentage of days with mean temperature above the assessment point at Reynolds Creek increases to 37%. This suggests that the site (all else being equal) may yet remain viable habitat for *L. tumana* and likely other similar water temperature sensitive taxa for a sizable percentage of August days. We

have not collected any *L. tumana* in our macroinvertebrate samples at the site; however, the species requires specialized identification (of adults; Giersch et al. 2015) and it is possible that some of our coarser identifications are *L. tumana*.

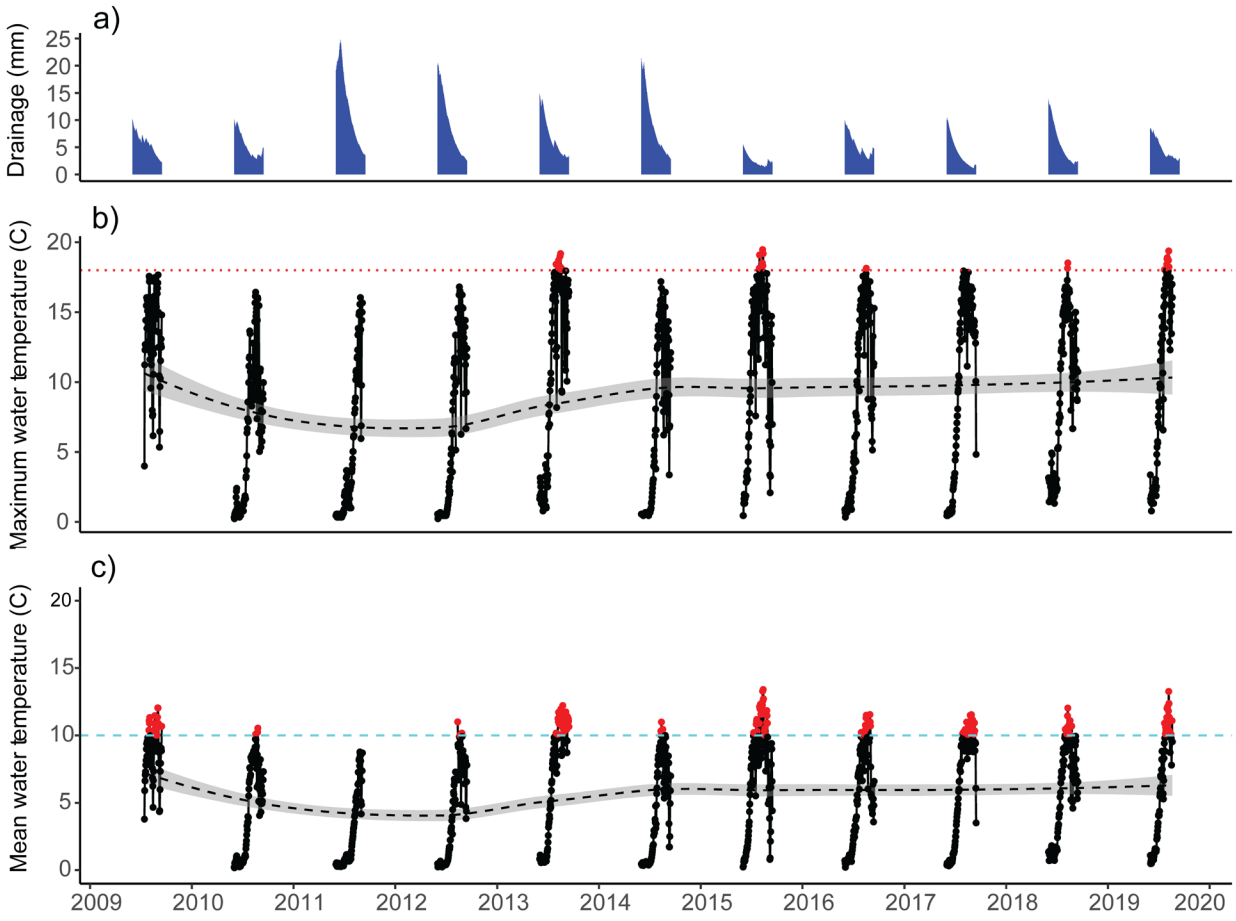


Figure 28. Daily stream temperature compared to assessment point at Reynolds Creek in Glacier National Park. The upper figure (a) gives drainage from a water balance model. The middle panel (b) shows daily maximum temperatures with a loess fit (black dashed line) and a 95% confidence interval (gray band). The red dotted line is a temperature assessment point for *Lednia tumana* of 18 °C (Muhlfeld et al. 2011). Summer temperatures above this threshold are colored red. The lower panel (c) shows summer daily mean temperature with a loess fit and its 95% confidence interval. The cyan dashed line is a mean temperature assessment point for *Lednia tumana* of 10 °C (Muhlfeld et al. 2011). Summer temperatures above this threshold are colored red.

3.3.3.6 Discussion and management applications: climate change

Of all the responses in the SEI protocol, water temperature may have the most direct and clearest connection to climate change. There is a strong linkage from the principal attributes of climate—temperature and precipitation—via the water cycle and its interaction with topoedaphic features, to water temperatures. As air temperature increases and precipitation becomes more variable, with a

cascade of subsequent changes in the water cycle, we see these changes manifest in water temperature.

Pederson et al. (2010) shows how, over the past century, air temperatures in the northern Rocky Mountains have warmed two to three times the rate of the global average. Isaak et al. (2012) and Crozier et al. (2008) present analyses of water temperature in the Pacific Northwest that show increases in the magnitude and duration of warm summer temperatures, with estimated increases up to 0.22 °C/decade and warmer temperatures beginning a full month earlier and persisting 2–3 weeks later. The decadal increases presented in Isaak et al. (2012) cover the period from 1980 to 2009 and include Glacier National Park. *Our models, beginning in 2009, suggest increases in water temperature have accelerated and are on the order of 1 to 1.5 ° C a decade.* Letcher et al. (2016) estimated peak temperature increases at a rate of 0.63 °C per decade in western Massachusetts and Kaushal et al. (2010) found warming rates for rivers and streams across the US of 0.07–0.77 °C per decade. These other estimates are somewhat dated and for other spatial extents; however, they are also much less dramatic than what we see at our sentinel sites in Glacier.

Future work by the network will include modeling of temperature data along climate change scenarios scaled to Glacier National Park. We believe that our incorporation of the water cycle into water temperature prediction is somewhat novel and will afford additional insight into these forecasts.

Future Stream Water Temperature

Spatial model-based prediction of water temperatures has been done by several authors at various scales, including the US Forest Service NorWest project (Isaak et al. 2012) and research applications by the USGS (Jones et al. 2014, 2017). Jones et al. (2014, 2017) presents models of water temperatures across space, back casting water temperature to streams without loggers and creating historical spatially explicit data. They then forecast temperatures across space and into the future using climate change scenario modeling. For our purposes we determined that the Jones et al. (2014, 2017) data was more useful.

Figure 29 presents USGS data (L. Jones, USGS hydrologist, pers. comm. 6/2015) that summarizes modeled climate change impacts on water temperature in the park. We subtracted baseline USGS modeled August temperatures from future predicted values across stream reach segments in the park. Jones et al. (2014, 2017) describes the process for modeling empirical temperature data (including a subset of the network logger data presented here) across space using a spatial flow-routed model of elevation, slope, aspect, latitude and longitude, solar radiation, air temperature, lake and glacier effects, and discharge. These were back cast to a baseline period of 1985 to 2005 (1995 midpoint). We use a future scenario developed in Jones et al. (2014, 2017) using an ensemble of climate models and an 8.5 representative concentration pathway (RCP) that estimates a relatively high emissions level from the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC 2013). We chose a future time period of 2026 to 2045 (2035 midpoint) as not too distant in the future and thus perhaps more management relevant.

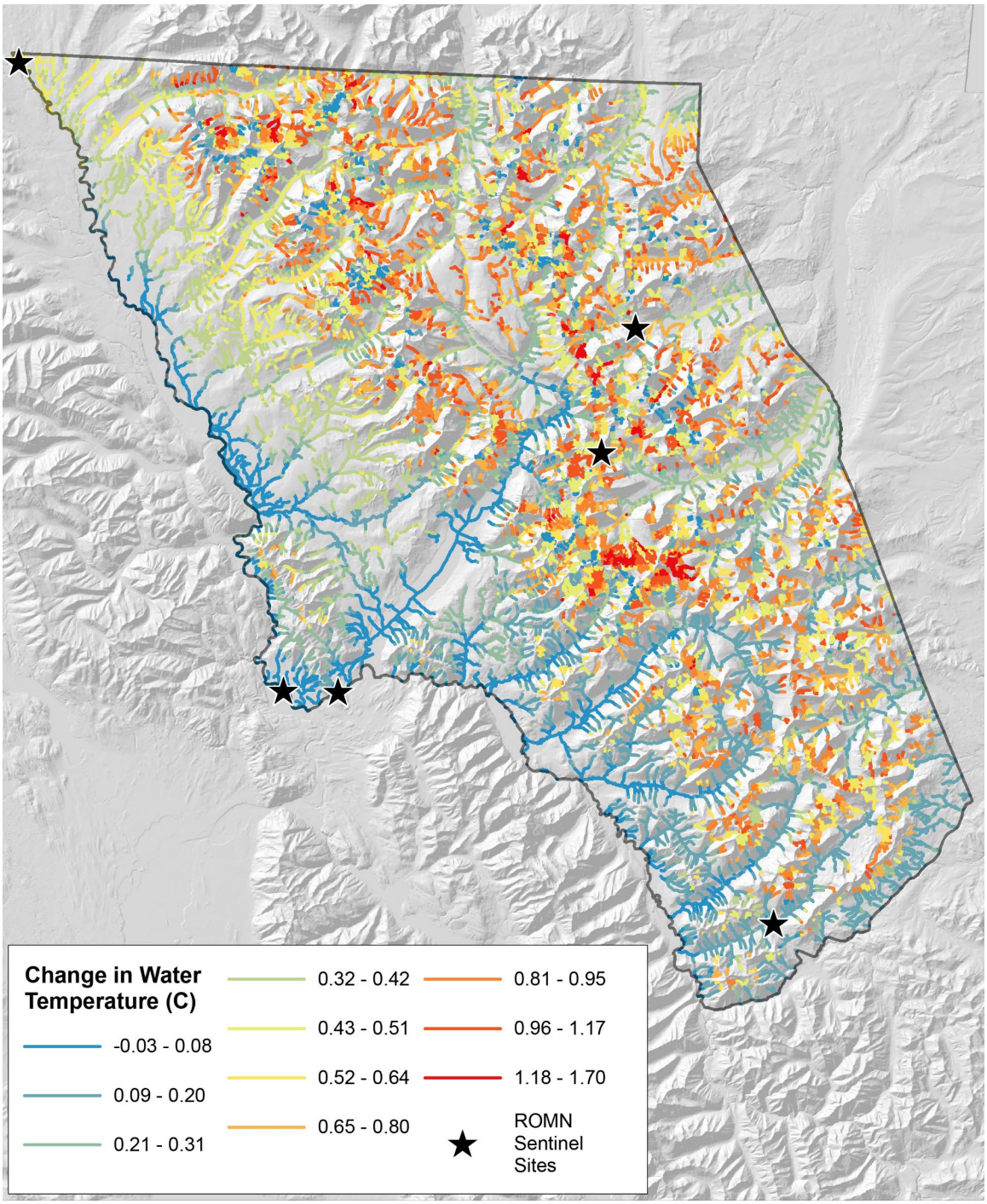


Figure 29. Projected changes in stream water temperatures at Glacier National Park. USGS (Jones et al. 2017) modeled reach-scale water temperature in a baseline period spanning 1985 to 2005 and predicted future temperatures in 2026 to 2045 using ensemble model predictions and a representative concentration pathway (RCP) 8.5 emissions scenario (IPCC 2013). Stars indicate Rocky Mountain Network (ROMN) stream monitoring sites. Note that blue colors include decreases, no change, and small increases in water temperature.

The USGS modeled stream temperature data suggest the parkwide mean increase was 0.52 °C over a 40-year interval from midpoints of 1995 to 2035, with a minimum change of −0.03 and a maximum change of 1.7 °C. Differences between baseline and future water temperatures have a clear spatial pattern across the park, with larger increases in alpine areas (a simple linear model suggests a significant increase with elevation, although the R^2 is small, at 0.18). The park's east side waters warmed more than the west side over this interval, again especially in alpine areas. Lower elevation streams on the park's west side have smaller increases or even very small projected decreases in August temperatures.

Patterns across most network sentinel sites fit into these modeled climate change driven scenarios, at least in a relative sense. Swiftcurrent Creek on the park's east side and at a mid-elevation had the largest temperature increase and most significant change over our period of record, pushing 2 °C over a decade. In a relative sense, this roughly matches the change in the USGS modeled data, although we see a greater increase in network data. Several authors (e.g., IPCC 2018; Tewksbury et al. 2011; Botha 2019) have noted increasing rates of changes in climate over the last decade, and our data, more recent than USGS data, may reflect this. Rocky Mountain Network sites on the park's west side on the Flathead River system and Ole Creek had smaller increasing trends or even small decreases, and this again matches smaller changes in the USGS data. Our site in the alpine on Reynolds Creek is more complex. It is located within a region of the park that has and will likely into the future experience the largest increases in water temperatures. Our data do suggest that the site became warmer by about 1.5 °C from 2009 to 2019, but we did not find this trend to be significant. These data were more difficult to model, with extreme diel fluctuations and a much shorter ice-free season. We may need a longer period of record to find a significant trend or perhaps a different metric, such as length of the ice-free period. We may also need to include the difference between night and day temperatures as an additional source of variation in any possible trend.

In summary, Rocky Mountain Network and partner data strongly suggest that climate change is impacting the park's waters, certainly via increased temperatures but also several other responses we present in this report. Our models link these changes to attributes of the water cycle and hydrologic processes that themselves are being rapidly modified by changing climate. For example, evidence suggests that evapotranspiration is increasing across broad scales (Walter et al. 2004) and this will impact water temperature. Water in Glacier National Park, as the name suggests, is influenced by Glacier's iconic features. As melt-induced flows are reduced by climate change (Huss and Hock 2018), streams, especially in the alpine, will become shallower, with less flow, elevating temperatures and the risk to important species.

3.3.4 In situ physiochemistry: dissolved oxygen, pH, and specific conductance

We first present three in situ water chemistry parameters collected via a short-term sonde deployment during each sample event: dissolved oxygen, pH, and specific conductance (Table 12). A fourth parameter, water temperature, is treated in more detail in the previous section.

Table 12. Select in situ parameters at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside (or in exceedance, if a regulatory criteria assessment point type) gives the percentage of results above (and/or below) an assessment point(s). Any Percent outside value above or equal to 25% is in bold. If more than one assessment point (AP) is used, the second AP is in parentheses. AP types include regulatory criteria (CRT), ecological threshold (THR), and ecoregional (ERG). Note that all parameters except dissolved oxygen increase as condition in a stream deteriorates (condition decreases as dissolved oxygen decreases). Values for all parameters except dissolved oxygen above or equal to the AP are in a non-reference condition (values for dissolved oxygen below the AP are in non-reference condition). For ecoregional assessment points (ERG), we use the 95th percentile value from 60 ecoregional reference sites.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside / in exceedance of AP	AP	AP type	Start date	End date	Sample size	Number of Sondes	2007–2009 park scale median	2007–2019 site scale median	Mean	Standard deviation	Min	Max
Dissolved oxygen, in water (mg/L)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	5(8)	CRT	8/15/2008	9/30/2019	7	2	9.300	10.740	10.626	0.935	9.61	11.61
	North Fork Flathead—Border	−0.1371	0.327	0.0	5(8)	CRT	7/11/2008	9/30/2019	43	2	9.300	10.780	10.582	0.803	8.92	11.96
	Swiftcurrent Creek	−0.0253	0.807	0.0	5(8)	CRT	10/18/2007	10/7/2019	44	3	9.300	10.470	10.034	1.021	7.86	11.69
	Reynolds Creek	0.0259	0.877	0.0	5(8)	CRT	9/11/2008	8/20/2019	30	2	9.300	9.375	9.340	1.093	7.71	10.97
	Ole Creek	−0.0048	0.962	0.0	5(8)	CRT	8/27/2009	10/2/2019	41	2	9.300	10.500	10.260	0.757	8.495	11.63
	Middle Fork Flathead	–[a]	–[a]	0.0	5(8)	CRT	8/28/2018	9/30/2019	6	1	9.300	11.000	10.858	0.481	10.14	11.27
pH, in water (s.u.)	North Fork Flathead—Glacier Rim	–[a]	–[a]	14.3	8.5(6.5)	THR	8/15/2008	9/30/2019	7	2	8.200	8.300	8.298	0.336	7.86	8.87
	North Fork Flathead—Border	0.0314	0.228	18.6	8.5(6.5)	THR	7/11/2008	9/30/2019	43	2	8.200	8.250	8.285	0.236	7.8	8.74

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend is significant ($p \leq 0.10$).

[c] Percent outside value is ≥ 25%.

Table 12 (continued). Select in situ parameters at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside (or in exceedance, if a regulatory criteria assessment point type) gives the percentage of results above (and/or below) an assessment point(s). Any Percent outside value above or equal to 25% is in bold. If more than one assessment point (AP) is used, the second AP is in parentheses. AP types include regulatory criteria (CRT), ecological threshold (THR), and ecoregional (ERG). Note that all parameters except dissolved oxygen increase as condition in a stream deteriorates (condition decreases as dissolved oxygen decreases). Values for all parameters except dissolved oxygen above or equal to the AP are in a non-reference condition (values for dissolved oxygen below the AP are in non-reference condition). For ecoregional assessment points (ERG), we use the 95th percentile value from 60 ecoregional reference sites.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside / in exceedance of AP	AP	AP type	Start date	End date	Sample size	Number of Sondes	2007–2009 park scale median	2007–2019 site scale median	Mean	Standard deviation	Min	Max
pH, in water (s.u.) (continued)	Swiftcurrent Creek	0.0069	0.860	2.3	8.5(6.5)	THR	10/18/2007	10/7/2019	44	3	8.200	7.940	7.917	0.421	6.85	8.74
	Reynolds Creek	0.0288	0.648	16.7	8.5(6.5)	THR	9/11/2008	8/20/2019	30	2	8.200	7.360	7.203	0.547	6.18	7.89
	Ole Creek	−0.0080	0.852	4.9	8.5(6.5)	THR	8/27/2009	10/2/2019	41	2	8.200	7.790	7.768	0.394	6.79	8.55
	Middle Fork Flathead	–[a]	–[a]	0.0	8.5(6.5)	THR	8/28/2018	9/30/2019	6	1	8.200	8.095	8.007	0.307	7.45	8.26
Specific conductance, in water (µs/cm)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	284	ERG	8/15/2008	9/30/2019	7	2	152.600	207.600	199.786	29.987	148.5	229.4
	North Fork Flathead—Border	4.8343	**0.001**[b]	0.0	284	ERG	7/11/2008	9/30/2019	43	2	152.600	228.900	229.298	41.393	163.6	282.8
	Swiftcurrent Creek	1.2472	0.224	0.0	284	ERG	10/18/2007	10/7/2019	44	3	152.600	112.350	115.332	11.005	97.04	138.5
	Reynolds Creek	0.1455	0.944	0.0	284	ERG	9/11/2008	8/20/2019	30	2	152.600	31.728	37.107	15.726	20.4	80.12

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend is significant ($p \leq 0.10$).

[c] Percent outside value is ≥ 25%.

Table 12 (continued). Select in situ parameters at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside (or in exceedance, if a regulatory criteria assessment point type) gives the percentage of results above (and/or below) an assessment point(s). Any Percent outside value above or equal to 25% is in bold. If more than one assessment point (AP) is used, the second AP is in parentheses. AP types include regulatory criteria (CRT), ecological threshold (THR), and ecoregional (ERG). Note that all parameters except dissolved oxygen increase as condition in a stream deteriorates (condition decreases as dissolved oxygen decreases). Values for all parameters except dissolved oxygen above or equal to the AP are in a non-reference condition (values for dissolved oxygen below the AP are in non-reference condition). For ecoregional assessment points (ERG), we use the 95th percentile value from 60 ecoregional reference sites.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside / in exceedance of AP	AP	AP type	Start date	End date	Sample size	Number of Sondes	2007–2009 park scale median	2007–2019 site scale median	Mean	Standard deviation	Min	Max
Specific conductance, in water (µs/cm) (continued)	Ole Creek	2.4087	0.177	0.0	284	ERG	8/27/2009	10/2/2019	41	2	152.600	94.325	86.015	22.718	51.15	125.4
	Middle Fork Flathead	–[a]	–[a]	0.0	284	ERG	8/28/2018	9/30/2019	6	1	152.600	183.450	181.525	28.997	135.7	215.15
Temperature, in water (C)	North Fork Flathead—Glacier Rim	–[a]	–[a]	**28.6**[c]	15	THR	8/15/2008	9/30/2019	7	2	9.600	10.930	10.373	4.046	5.73	15.41
	North Fork Flathead—Border	0.4561	0.315	0.0	15	THR	7/11/2008	9/30/2019	43	2	9.600	5.830	6.919	2.852	3.07	12.88
	Swiftcurrent Creek	0.5148	0.426	11.4	15	THR	10/18/2007	10/7/2019	44	3	9.600	5.240	7.362	4.954	1.04	19.3
	Reynolds Creek	−0.1734	0.839	3.3	15	THR	9/11/2008	8/20/2019	30	2	9.600	6.783	7.165	4.881	0.36	15.505
	Ole Creek	−0.3504	0.302	0.0	15	THR	8/27/2009	10/2/2019	41	2	9.600	5.990	6.912	2.791	2.805	13.11
	Middle Fork Flathead	–[a]	–[a]	0.0	15	THR	8/28/2018	9/30/2019	6	1	9.600	8.440	8.592	1.531	6.84	10.99

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend is significant ($p \leq 0.10$).

[c] Percent outside value is ≥ 25%.

3.3.4.1 Dissolved oxygen

Background and relevancy

Dissolved oxygen (DO) is the amount of oxygen in solution measured as either a percentage or in mg/L (Figure 30). DO is closely linked to physical and biological processes within streams and is necessary for the survival of many aquatic organisms. In general, there may be a connection between reduced DO and lower biodiversity (i.e., Verberk et al. 2011). DO is created and depleted by chemical and biological processes and via atmospheric exchange driven by turbulence in rapids and riffles. In addition to high water temperatures, high microbial activity—driven by organic pollution—drives demand for dissolved oxygen, which can result in reduced or anoxic conditions. High oxygen levels are especially critical for the metabolism of aquatic insects and salmonid eggs. Dissolved oxygen is important to park management because, as just one example, it can impact fish range and habitat quality.



Figure 30. Glacier National Park crews collect in situ dissolved oxygen samples on Swiftcurrent Creek in the park, 2011. NPS / J. GUSE

Assessment points

The State of Montana (Montana DEQ 2002) establishes a minimum instantaneous DO concentration of 8.0 mg/L for early life stages of salmonids in class A-1 and Outstanding Resource Water (ORW)

streams, such as in Glacier National Park. DO levels *below* these assessment points are indicative of non-reference conditions. Montana DEQ applies these criteria to instantaneous synoptic data, such as the type of data we collect in SEI sentinel monitoring (Montana DEQ 2017).

Parkwide 2007–2009 baseline and other previous results

Park-scaled unbiased results for DO in the 2007–2009 baseline survey were variable, likely due to high levels of diel and seasonal variation. However, overall, DO levels in Glacier in 2007–2009 did not suggest meaningful disturbance or systemic problems in the park's streams, with a median of 9.3 mg/L. Other work with DO has shown similar results. Mast (2007) presents site specific data from 1984 to 2004, with median DO levels at 10.2 mg/L. Mills et al. (2011) reported a site-specific median value from 2007 to 2008 for the North Fork Flathead—Border site of 11.1 mg/L in 2007–2008.

Trend

No site had a statistically significant or even marginal trend (Table 12, Figure 31). There was no indication of any sonde effect in DO results. There was a strong negative effect of flow at Ole Creek, with lower DO at lower flows. We suspect a model that included a temperature covariate might further improve this relationship. Season was also important at Ole Creek, with higher oxygen levels during spring runoff. Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites.

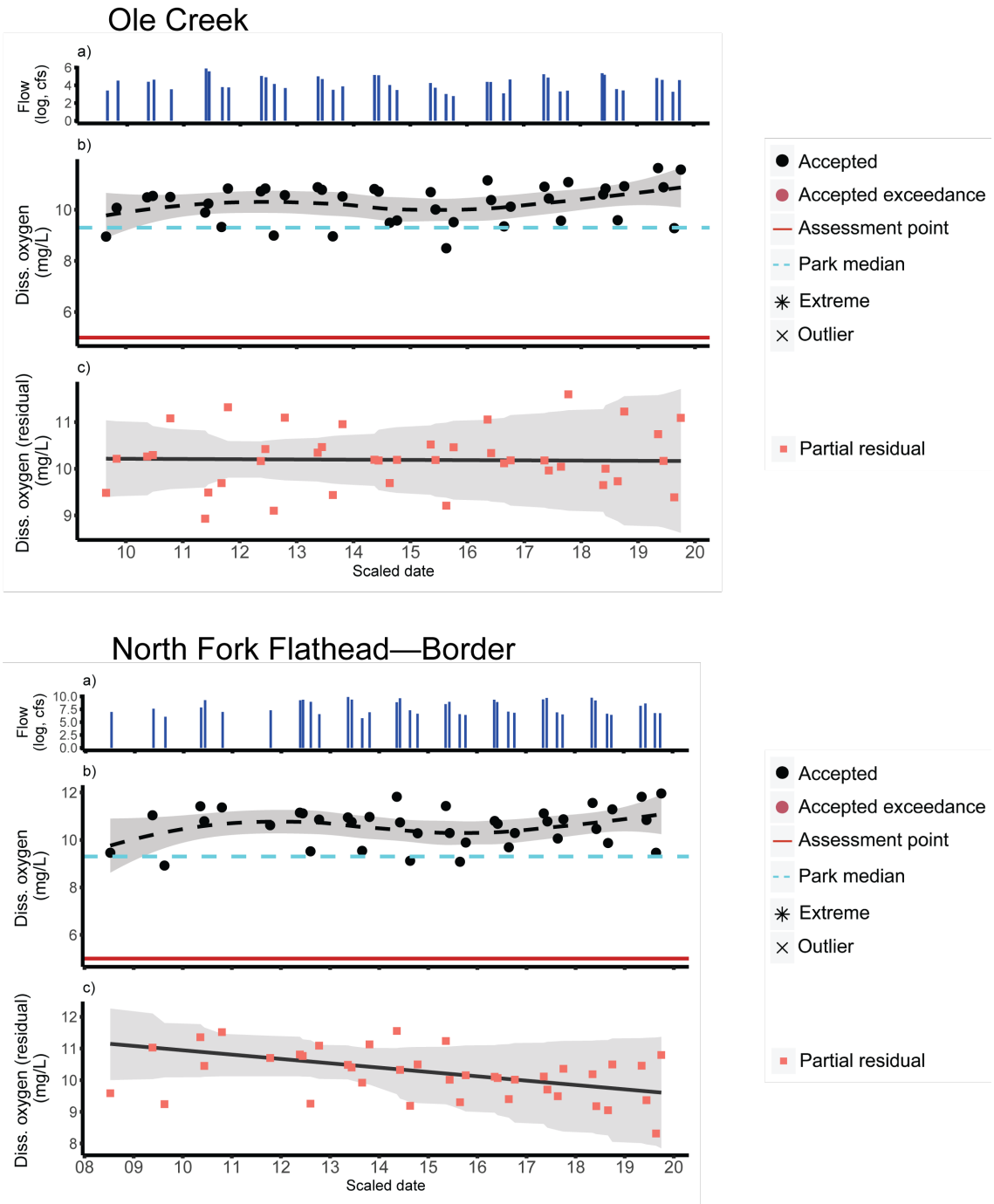


Figure 31. Select trends in dissolved oxygen (mg/L) in Glacier National Park from 2008 to 2019 at Ole Creek (top) and North Fork Flathead—Border (bottom) sentinel sites. (a) Stream flow at each sample event, as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event, with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band); as available, assessment point (red solid line) and 2007–2009 park-scale median are shown.(c) Partial residuals of the response vs. scaled date from the main quantile regression model; the fitted line, (not significant for this metric), and its pointwise 95% confidence interval (light gray band) are from a quantile regression model at the median of the residual value vs. scaled date. Legend contains symbols that may not apply to this figure.

Condition

No dissolved oxygen result was outside the assessment point at any site over our period of record (Table 12; Figure 31). Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead had median DO values like other large river sentinel sites and no values were outside the assessment points.

Discussion and management applications

Dissolved oxygen levels are driven by several factors, including temperature, flow, and nutrient concentrations, all of which can vary with levels of human disturbance. Warmer water reduces the amount of oxygen that can be dissolved and used in biological processes in a stream (IPCC 2018). Ficklin et al. (2013) show that with a spring and summer stream temperature increase of 1.5 to 5.5 °C, a decrease in DO of 10% by 2100 might be expected. As presented below, we see an increase in water temperatures in the park of around 1.5 °C per decade. Therefore, while our DO data did not reveal a drop in levels at sentinel sites from 2007 to 2019, as water temperatures continue to increase, DO levels should be watched closely.

3.3.4.2 pH

Background and relevancy

pH measures hydrogen ion concentration in fluids, indicating how acidic or basic a fluid is, with the midpoint 7, being neutral. pH has many important physical and biological effects in aquatic systems. Because most aquatic species occur within specific habitat envelopes of pH conditions based on their physiology, changes in pH affect their physiological tolerances and thus may ultimately result in changes in species assemblages. pH determines the solubility and biological availability of many nutrients and heavy metals in water. For instance, most metals become more soluble at lower pH levels, and some of those become toxic to aquatic organisms.

Assessment points

The State of Montana establishes standards for pH in ORW waters. The standards indicate that any induced (i.e., anthropogenic) variation of pH within the range of 6.5 to 8.5 must be less than 0.5 pH units, and that natural pH outside this range must be maintained without change. In addition, natural pH above 7.0 must be maintained above 7.0 (Montana DEQ 2002). We use 6.5 and 8.5 as lower and upper assessment points (but with caveats as discussed below).

Parkwide 2007–2009 baseline and other previous results

Park-scaled unbiased results for pH in the 2007–2009 baseline survey were variable, with a median of 8.2. There were some values outside the assessment points; however, there was no evidence that any pH exceedance was “unnatural.” Previous work on pH by network partners had similar results. Mast (2007) analyzed site specific data from 1984 to 2004, with median pH levels at 8.3. Mills et al. (2011) reported a site-specific median value from 2007 to 2008 for the North Fork Flathead—Border of 8.2. Both reports also showed no trend in pH at specific sites.

Trend

The direction of change across time in pH was positive at all sites except Reynolds Creek. However, no site had a significant or marginally significant trend (Table 12, Figure 32). There was no

indication of any sonde effect in any pH trend model. Flow did not play a significant role in any pH model. Season was significant at two sites with more basic (i.e., higher) pH values in the fall. Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites.

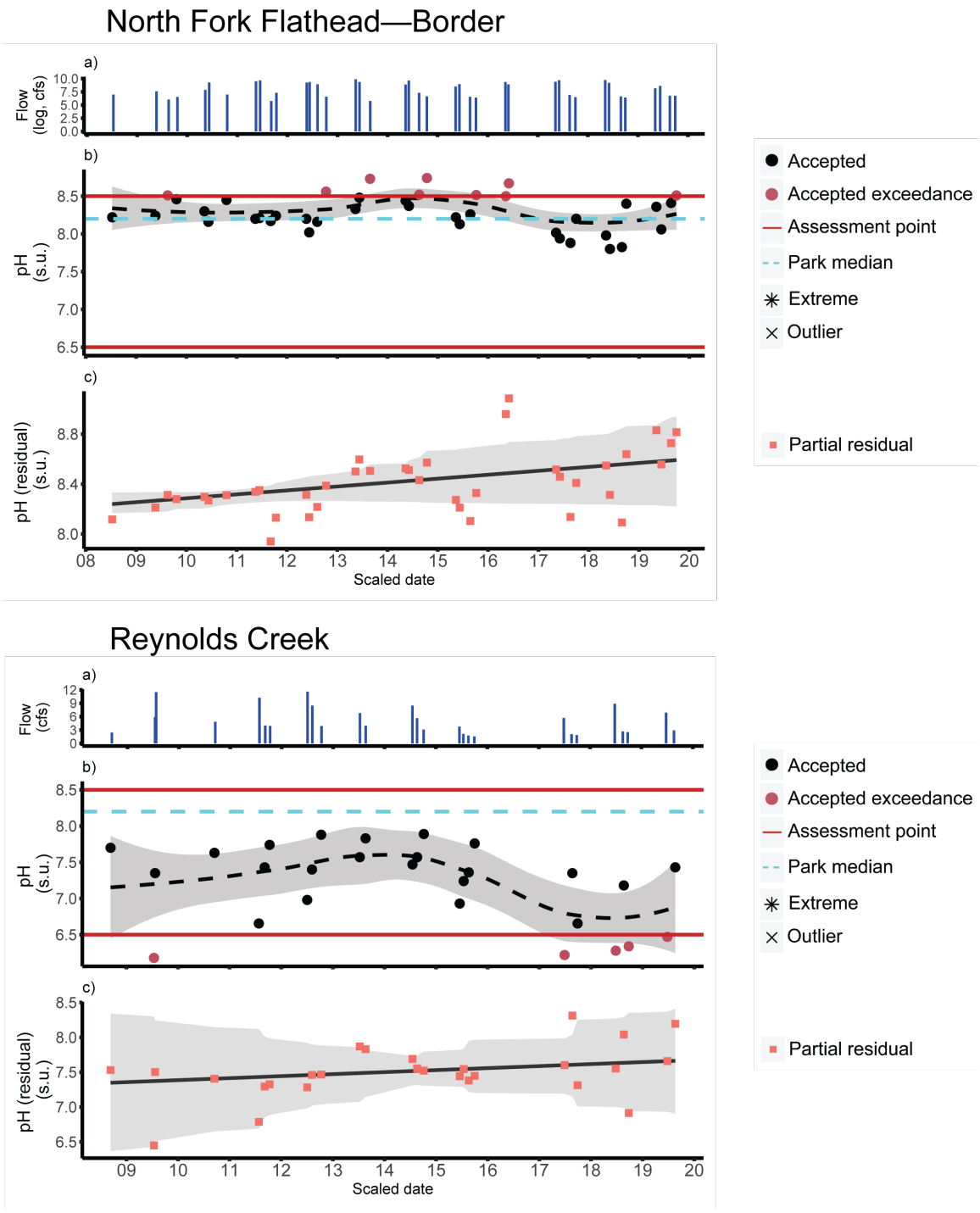


Figure 32. Select trends in pH (s.u.) at sentinel stream monitoring sites in Glacier National Park from 2008 to 2019: (top) North Fork Flathead—Border and (bottom) Reynolds Creek. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment points (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (not significant for this metric) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

Condition

Five of the six sites had values outside of the Montana DEQ assessment points for pH over our period of record. Percent exceedances ranged from 2.2% to 18.6% and included results on the acidic side (Reynolds Creek) and basic side (all other sites). While some of these percentages of values outside of the ideal pH range defined by Montana DEQ seem high for a wilderness park, to be a regulatory issue, a connection to manipulation of pH would be necessary and this does not exist in the park. pH in exceedance in the park does not represent any direct anthropogenically induced variation as required by the state's criteria to be of concern. There are complex interactions with other water chemistry parameters that can influence pH, some of which may have anthropogenic sources upstream or, in select cases, within Glacier. None of these exceedances are indicative of a pH problem.

Median pH values at sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead are similar to other pH results; however, 14.2% of the readings at Glacier Rim were slightly above the upper assessment point. There is some recreational use of this site and limited development in the watershed. However, there is no clear indication that these led to any change in pH values at the site, and with a sample size of only eight over two years, this is not likely meaningful.

Discussion and management applications

Most streams in Glacier are poorly buffered or have naturally low alkalinity and they are susceptible to pH increases (acidification) from both dry and wet deposition of sulfate and nitrogen oxides. Sulfate emissions have been reduced in North America over the past 20 years but still exceed preindustrial levels. Regional nitrogen oxide emissions have been either stable or increasing (Kuylenstierna et al. 2001). Alpine streams, like our Reynolds Creek site, have the highest risk of acidification (i.e., Skjelkvale and Wright 2001). Reynolds Creek had notably more acidic conditions (and lower alkalinity, see below) than the park median or other sentinel sites, and several readings late in our period of record were below the lower assessment point for pH. However, our data do not show a significant trend of decreasing pH (acidification) at the site. So perhaps, at least over our relatively short period of record, pH was not problematic in Reynolds Creek. Yet alpine catchments are often stoichiometrically inert, with extremely dilute pristine waters (see Ellis et al. 1992 for a discussion along these lines for remote lakes), and if an alpine stream acidifies, it can leach metals (Al, Cd, Zn, Mn, etc.) from soils in their watershed (and we may be seeing this, see below). This can lead to abrupt drops in pH in streams during runoff events. This pH shock can be dangerous for biological systems (Moiseenko 2005; Lepori et al. 2003) and monitoring should continue to look for this.

3.3.4.3 Specific conductance

Background and relevancy

Specific conductance (SC) is the ability of a water body to conduct an electric current. Specific conductance is used in stream monitoring as an indicator of the dissolved ion concentration, as this is directly correlated with current conductance. The more dilute the water, the lower the concentrations of dissolved ions, and thus, the lower the conductance. Changes in conductance can indicate changes

in potassium, calcium, or other anions and cations. No change in conductance over time generally means major ion concentrations are stable.

Assessment points

There are no relevant Montana DEQ or EPA criteria for SC. Our assessment point of 284 µS/cm is an ecoregional assessment point derived from values at reference sites from across the ecoregions Glacier is embedded within (Schweiger et al. 2018). It has no regulatory authority; rather it estimates the upper limits of SC in comparable streams sites that are in reference condition and provides context for results in Glacier under an expectation of a reference state.

Parkwide 2007–2009 baseline and other previous results

Results in the 2007–2009 baseline survey were variable, likely due to the high levels of diel and seasonal variation in SC. Specific conductance levels in Glacier in 2007–2009 did not suggest meaningful disturbance or systemic problems in the park's streams due to ion concentrations. The population scale median of 152.6 µS/cm was well below the ecoregional assessment point. Some baseline results in larger rivers like the North Fork of the Flathead approached or occasionally exceeded the assessment point, but larger rivers have higher SC naturally, and with no direct anthropogenic cause, this was not considered problematic. Mast (2007) reported similar site-specific data from 1984 to 2004 with a median of 101µS/cm, and Mills et al. (2011) reported a site-specific median value from 2007 to 2008 for the North Fork Flathead—Border of 238 µS/cm. Both reports also showed some small but significant and positive trends in SC at specific sites on the North Fork of the Flathead. These increasing trends cover a period of record spanning the late 1970s to the mid-2000s and thus predate our own trend models.

Trend

All sites had increasing trends, with the North Fork Flathead—Border strongly significant and Ole Creek marginally significant (Table 12, Figure 33). There was no indication of any sonde effect in the specific conductance trend model. Season was significant at all sites, with higher values in the fall and early winter when flows are reduced and the relative concentration of the dissolved solutes that elevate conductance were higher. Flow did play a significant role at two of the four trend sites, but with mixed patterns in its effect.

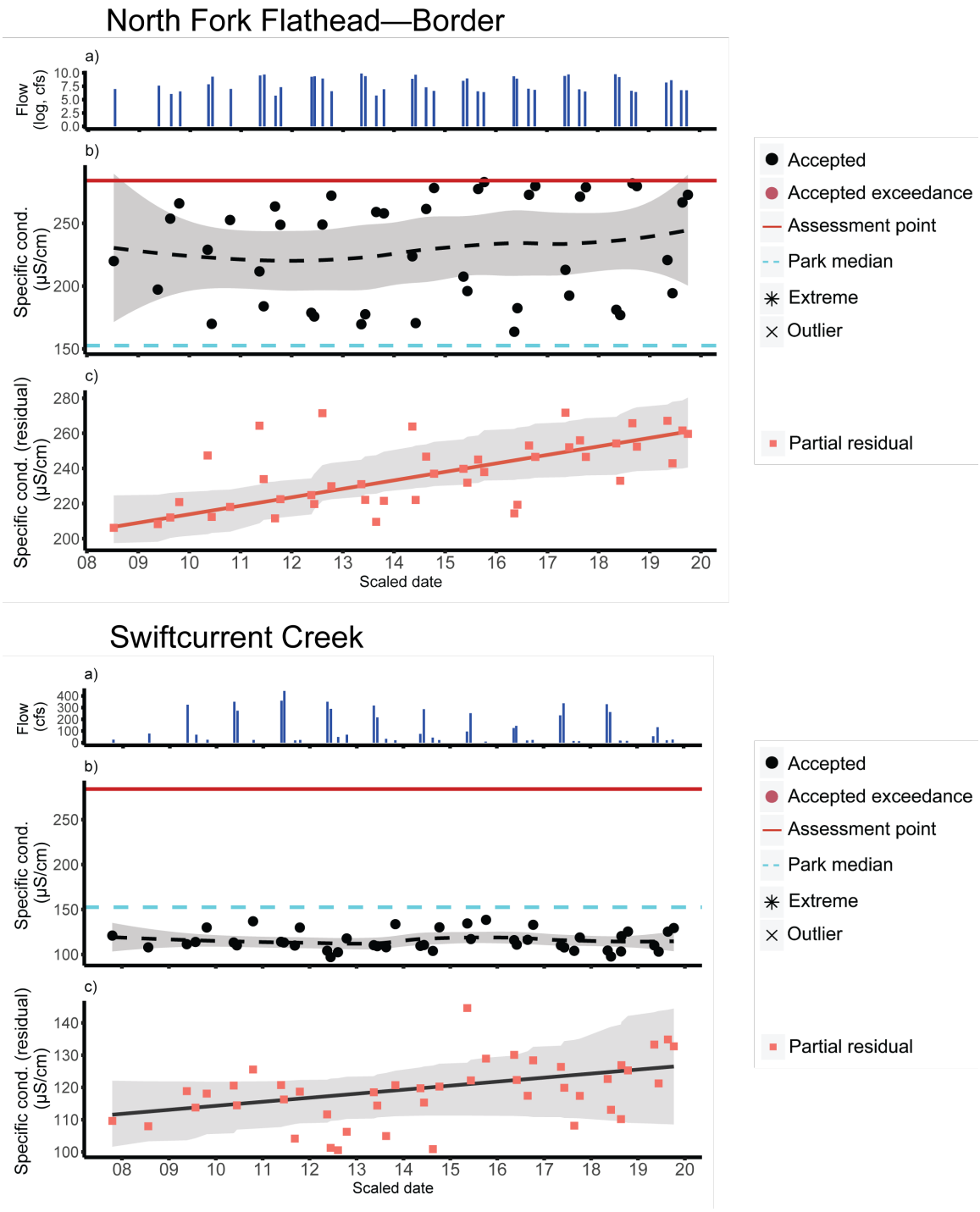


Figure 33. Select trends in specific conductance (µS/cm) at sentinel stream monitoring sites in Glacier National Park from 2008 to 2019: (top) North Fork Flathead—Border and (bottom) Swiftcurrent Creek. a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment point (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (significant for North Fork Flathead—Border site) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites.

Condition

No specific conductance value was greater than our ecoregional assessment point, suggesting that, across time, Glacier sentinel stream sites had SC below the best sites in the surrounding ecoregions (Table 12, Figure 33). Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead had median SC values like other large river sentinel site results and no values were outside the assessment points.

Discussion and management applications

The increasing trend at the North Fork Flathead—Border site extends similar trends seen in Mast (2007) and Mills et al. (2011). Like in our models, flow must be included in these models to estimate a significant trend. These authors also account for season, although they do not report how it impacts SC—in our models a seasonal term was required. They also do not include any test of a sonde effect, and while the summary statistics reported are median values, the trend models used mean values that are likely influenced by non-normal distributions and outliers. Nevertheless, these important details aside, it is interesting that two older reports, and now this publication, suggest increasing SC in the North Fork of the Flathead as this important river crosses the Canadian border and enters the park.

Maximum SC in our data, the historical reports, and where this trend is headed are very close to the ecoregional assessment point, suggesting that SC is nearing the limits of what might be considered a reference state on the North Fork of the Flathead. Work in landscapes with a more pervasive human footprint (i.e., the urbanizing coastal zone of New Jersey, Conway 2007) shows a clear relationship between higher levels of human disturbance and elevated SC. Other studies link SC driven by intense land use, like coal extraction, to a reduced biological condition. For example, Boehme (2013) shows a strong relationship between elevated SC and reduced integrity in Appalachian macroinvertebrate communities. We caution that these relationships are not diagnostic, that our assessment points do not have any regulatory authority, and more research is needed to account for likely many correlated factors in these relationships. However, because SC increases at all our sentinel sites (although only significantly at the North Fork Flathead—Border site) we suggest that, should this be confirmed with more work, some caution might be warranted.

3.3.5 Major ions and suspended sediment

Major ions (such as Calcium Ca++, Sodium Na+, Chloride Cl-, Sulfate SO4=), are chemical constituents (atoms or compounds) that carry a positive or negative charge important in the chemistry of stream and river water. Major oppositely charged ions often combine into neutralized “salts” that may also be critical in stream water chemistry. For example, magnesium chloride, formed from the light metal magnesium and chloride is in widespread use as a road deicer and thus is potentially important in effects to streams from runoff of roads. In this section, we first focus on the anions sulfate and chloride and the cation calcium. We then summarize alkalinity as another important combination of ions, followed by total suspended sediment, given the relationship between sediment and major ion concentrations.

3.3.5.1 Chloride, sulfate, and calcium

Background and relevancy

Chloride, sulfate, and calcium are all important ions in stream water. They have several natural sources, including dissolution from crystalline bedrock and minerals (e.g., gypsum), oxidation of pyrite, and surface or near-surface discharge from Paleozoic brines (Dyson 1953). Mast (2007) suggests that natural sources of chloride and sulfate, such as weathering of sedimentary rocks, is the primary pathway of these ions into the waters of the park. However, these anions can also be indicators of anthropogenic disturbance when elevated (Peters 1984; Biggs et al. 2004), reflecting contributions from a wide range of stressors, including agriculture, mining, and urban land use. There are no known or likely meaningful industrial or resource extraction point sources for these constituents in or close to the park; however, they can be volatilized from distant external human sources and distributed to streams and rivers in the park via atmospheric deposition (Landers et al. 2010). Chloride is also often used in salts applied to roads near the park boundary (Thunqvist 2004; Kelly et al. 2007) and in select and limited locations in the park (Williams 2003; C. Downs, NPS biologist, pers. comm. 12/2022) with excess potentially running off into streams. Sulfate can be released from oxidation of sulfide minerals in mine wastes and tailings ponds outside the park. Should the long-debated mining ever occur in the Canadian portion of the North Fork of the Flathead, this river might yet be impacted by sulfate (among other analytes). At least in other systems, there is good evidence these ions can have strong impacts on macroinvertebrate communities (Blasius and Merritt 2002) and other aspects of stream condition.

Assessment points

Montana DEQ and the US EPA have been developing modern guidance for chloride criteria, but as of December 2023, there were no final rulings. A dated US EPA criterion (EPA 2009) for chloride based on traditional toxicity to plant, fish, and invertebrate species in more human dominated landscapes is likely not ecologically relevant in the undisturbed watersheds of Glacier (i.e., its value of 230 mg/L is over three orders of magnitude greater than median chloride concentrations at our sites in the park). When suitable criteria become available, we will apply these to SEI data as assessment points. With no regulatory or park management assessment points or known ecological thresholds, we estimated ecoregional assessment points for chloride, sulfate, and calcium (Table 13).

Table 13. Select major ions at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s). Any percent outside value above or equal to 25% is in bold. All assessment point (AP) types are ecoregional in form. All parameters increase as condition in a stream deteriorates. Values above or equal to the assessment point are in a non-reference condition. For ecoregional assessment points, we use the 95th percentile value from 60 ecoregional reference sites.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside of AP	AP	Start date	End date	N	Percent censored	Number of labs	Number of detection limits	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Alkalinity as CaCO3, total in water (mg/L)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	144(10)	10/4/2007	9/30/2019	8	0.0	2	2	61.500	88.594	90.852	22.101	65.51	117.41
	North Fork Flathead—Border	1.0440	0.177	24.4	144(10)	8/20/2007	9/30/2019	45	0.0	3	3	61.500	122.638	119.480	24.299	76.01	153.26
	Swiftcurrent Creek	0.5567	0.883	0.0	144(10)	10/18/2007	10/7/2019	45	0.0	3	3	61.500	53.300	54.333	4.153	48.116	64.965
	Reynolds Creek	1.1207	0.785	9.4	144(10)	9/11/2008	8/20/2019	32	0.0	3	3	61.500	14.691	17.249	6.759	8.443	31.762
	Ole Creek	0.1113	0.450	0.0	144(10)	8/27/2009	10/2/2019	42	0.0	2	2	61.500	40.328	41.048	10.155	24.265	58.77
	Middle Fork Flathead	–[a]	–[a]	0.0	144(10)	8/28/2018	9/30/2019	6	0.0	1	1	61.500	67.335	73.048	17.346	53.45	97.23
Calcium, dissolved in water (mg/L)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	1840.7	N/A[b]	N/A[b]	*	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]
	North Fork Flathead—Border	0.1158	0.692	0.0	1840.7	5/21/2009	6/5/2017	33	0.0	1	7	16.1	38.130	35.983	7.193	25.01	45.274
	Swiftcurrent Creek	0.2478	**0.022**[c]	0.0	1840.7	5/20/2009	6/6/2017	33	0.0	1	7	16.1	14.990	15.163	1.425	12.92	18.102

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Calcium was not analyzed in 2018 and beyond

[c] Trend is significant ($p \leq 0.10$).

[d] Percent outside value is above or equal to 25%.

Table 13 (continued). Select major ions at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s). Any percent outside value above or equal to 25% is in bold. All assessment point (AP) types are ecoregional in form. All parameters increase as condition in a stream deteriorates. Values above or equal to the assessment point are in a non-reference condition. For ecoregional assessment points, we use the 95th percentile value from 60 ecoregional reference sites.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside of AP	AP	Start date	End date	N	Percent censored	Number of labs	Number of detection limits	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Calcium, dissolved in water (mg/L) (continued)	Reynolds Creek	0.2778	0.320	0.0	1840.7	7/15/2009	10/4/2016	23	0.0	1	7	16.1	3.864	5.025	2.277	2.615	9.938
	Ole Creek	0.1454	0.391	0.0	1840.7	8/27/2009	10/5/2016	30	0.0	1	7	16.1	10.001	9.442	2.347	5.957	13.373
	Middle Fork Flathead	–[a]	–[a]	0.0	1840.7	N/A[b]	N/A[b]	*	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]	N/A[b]
Chloride, dissolved in water (mg/L)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	0.77	10/4/2007	9/30/2019	8	0.0	2	3	0.300	0.315	0.358	0.129	0.21	0.58
	North Fork Flathead—Border	−0.0011	0.939	0.0	0.77	8/20/2007	9/30/2019	46	0.0	3	11	0.300	0.096	0.156	0.140	0.048	0.662
	Swiftcurrent Creek	0.0011	**0.019**[c]	0.0	0.77	10/18/2007	10/7/2019	45	2.2	3	11	0.300	0.060	0.083	0.079	0.033	0.476
	Reynolds Creek	0.0037	0.567	0.0	0.77	9/11/2008	8/20/2019	32	3.1	3	10	0.300	0.025	0.049	0.056	0.017	0.3
	Ole Creek	0.0018	0.530	0.0	0.77	8/27/2009	10/2/2019	42	0.0	2	9	0.300	0.073	0.091	0.045	0.045	0.225
	Middle Fork Flathead	–[a]	–[a]	0.0	0.77	8/28/2018	9/30/2019	6	0.0	1	1	0.300	0.515	0.500	0.112	0.3	0.64

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Calcium was not analyzed in 2018 and beyond

[c] Trend is significant ($p \leq 0.10$).

[d] Percent outside value is above or equal to 25%.

Table 13 (continued). Select major ions at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s). Any percent outside value above or equal to 25% is in bold. All assessment point (AP) types are ecoregional in form. All parameters increase as condition in a stream deteriorates. Values above or equal to the assessment point are in a non-reference condition. For ecoregional assessment points, we use the 95th percentile value from 60 ecoregional reference sites.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside of AP	AP	Start date	End date	N	Percent censored	Number of labs	Number of detection limits	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Sulfate as SO4, dissolved in water (mg/L)	North Fork Flathead—Glacier Rim	–[a]	–[a]	**25.0**[d]	8.6	10/4/2007	9/30/2019	8	0.0	2	3	2.800	7.552	6.531	2.880	1.287	9.736
	North Fork Flathead—Border	−0.1023	0.904	0.0	8.6	8/20/2007	9/30/2019	46	0.0	3	11	2.800	3.894	3.759	1.062	1.47	6.288
	Swiftcurrent Creek	0.0472	0.233	0.0	8.6	10/18/2007	10/7/2019	45	0.0	3	11	2.800	2.260	2.705	1.233	1.258	6.754
	Reynolds Creek	0.0456	0.544	0.0	8.6	9/11/2008	8/20/2019	32	0.0	3	10	2.800	0.987	1.329	0.965	0.445	4.185
	Ole Creek	0.0308	0.728	0.0	8.6	8/27/2009	10/2/2019	42	0.0	2	9	2.800	2.249	2.315	0.581	1.516	3.241
	Middle Fork Flathead	–[a]	–[a]	0.0	8.6	8/28/2018	9/30/2019	6	0.0	1	1	2.800	4.149	4.129	0.732	2.876	4.883
Total suspended solids, total in water (mg/L)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	6.7	10/4/2007	9/30/2019	8	62.5	2	2	0.800	0.081	0.879	1.685	0.026	4.94
	North Fork Flathead—Border	−0.1921	0.672	**33.3**[d]	6.7	8/20/2007	9/30/2019	27	29.6	3	5	0.800	1.432	17.791	34.618	0.027	150.99
	Swiftcurrent Creek	0.2608	0.915	0.0	6.7	10/18/2007	10/7/2019	26	42.3	3	5	0.800	0.470	0.843	1.065	0.051	4.6

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Calcium was not analyzed in 2018 and beyond

[c] Trend is significant ($p \leq 0.10$).

[d] Percent outside value is above or equal to 25%.

Table 13 (continued). Select major ions at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s). Any percent outside value above or equal to 25% is in bold. All assessment point (AP) types are ecoregional in form. All parameters increase as condition in a stream deteriorates. Values above or equal to the assessment point are in a non-reference condition. For ecoregional assessment points, we use the 95th percentile value from 60 ecoregional reference sites.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside of AP	AP	Start date	End date	N	Percent censored	Number of labs	Number of detection limits	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Total suspended solids, total in water (mg/L) (continued)	Reynolds Creek	0.3150	0.840	0.0	6.7	9/11/2008	8/20/2019	18	61.1	3	5	0.800	0.418	0.475	0.270	0.044	1.244
	Ole Creek	−0.0542	0.901	0.0	6.7	6/11/2014	10/2/2019	22	40.9	2	4	0.800	0.701	1.261	1.280	0.06	4.75
	Middle Fork Flathead	–[a]	–[a]	0.0	6.7	8/28/2018	9/30/2019	6	66.7	1	1	0.800	0.963	1.905	1.534	0.862	4.17

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Calcium was not analyzed in 2018 and beyond

[c] Trend is significant ($p \leq 0.10$).

[d] Percent outside value is above or equal to 25%.

Parkwide 2007–2009 baseline and other previous results

Chloride, sulfate, and calcium concentrations were relatively low in Glacier in 2007–2009, with park scale medians of 0.3, 2.8 and 16.1 mg/L, respectively. From zero to a small percentage of the park's streams were above ecoregional assessment points, usually at sites near roads or front country visitor facilities. Mast (2007) reported site-specific data from 1980 to 2004 with similar medians to our park-scale baselines of 0.1, 2.3 and 41 mg/L for chloride, sulfate, and calcium, respectively. Mills et al. (2011) reported a site-specific median value for the North Fork Flathead—Border from 2007 to 2008 at 0.12, 4.1 and 37.1 mg/L for chloride, sulfate, and calcium, respectively. Both reports also showed some small and significant negative trends in sulfate and positive trends in calcium at the North Fork Flathead—Border site from the early 1970s to the mid-1990s. Mills et al. (2011) suggest that there may be an unidentified sulfate source in the northern tributaries to the park because sulfate loads varied with concentration, not stream flow. The trend coefficients for chloride in both of these reports were very small at all sites and were not significant (Figure 34).

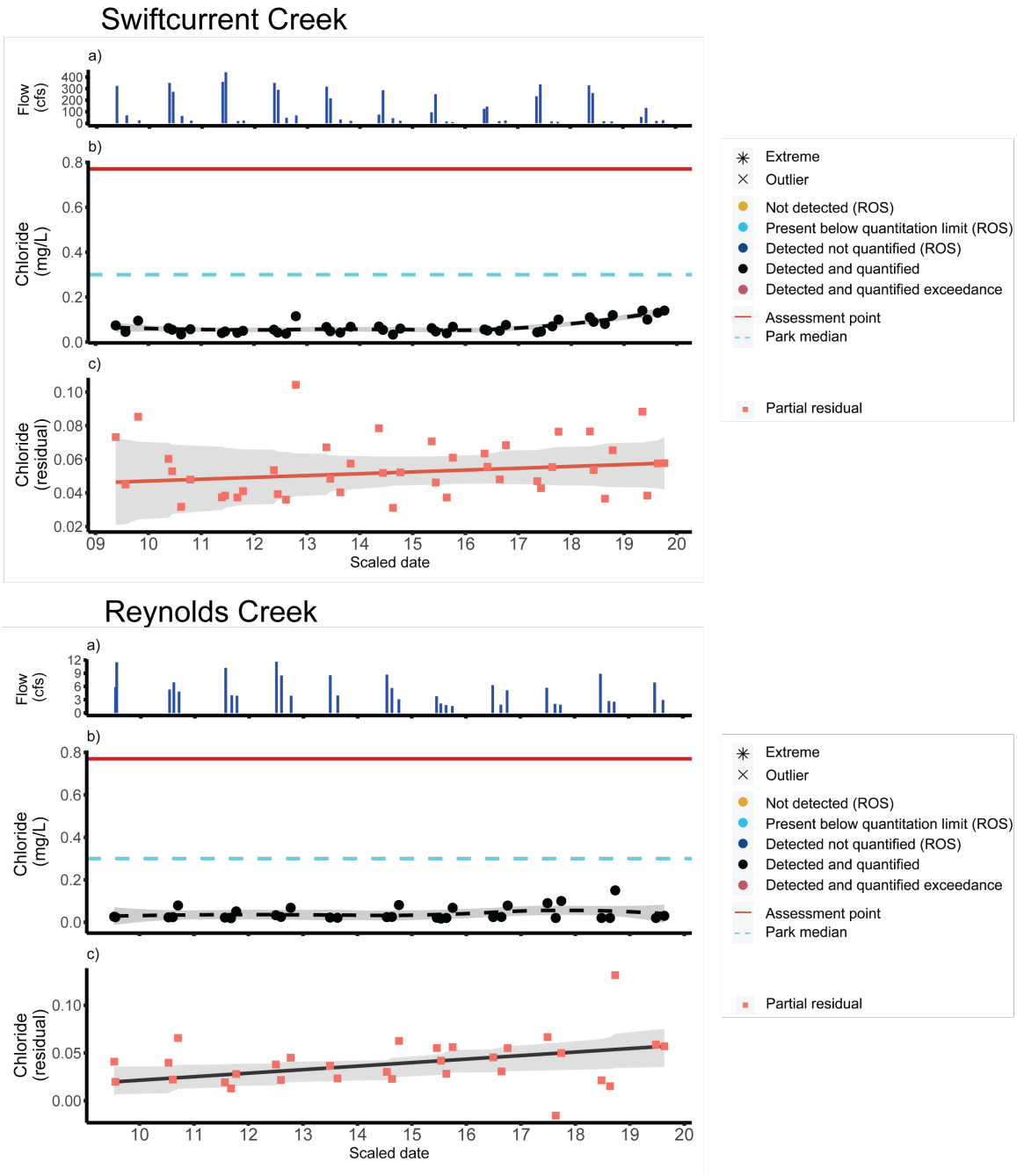


Figure 34. Select trends in chloride (mg/L) in Glacier National Park from 2009 to 2019 at (top) Swiftcurrent Creek and (bottom) Reynolds Creek. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment point (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (significant for the Swiftcurrent Creek site) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

Trends

Calcium, sulfate, and chloride levels changed little from 2007 to 2019 at most sentinel stream sites; coefficients for trend terms in the quantile regression models were both positive and negative, but all were small. Two exceptions were chloride and calcium, both at Swiftcurrent Creek, with significant increasing trends. Table 13 summarizes key statistics and results for sulfate, chloride, and calcium at sentinel sites (Figure 35, 36). Chloride at Swiftcurrent Creek had a lab effect that was statistically removed in the quantile regression model. Season was significant for chloride at Reynolds Creek, with higher values in the fall and early winter, where flows were much reduced and the relative concentrations of ions can increase. Flow was significant for chloride and calcium at the North Fork Flathead—Border, with higher concentrations at lower flows. During high-flow conditions, the influx of dilute snowmelt water likely drives an overall decline in concentrations of major ions.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites.

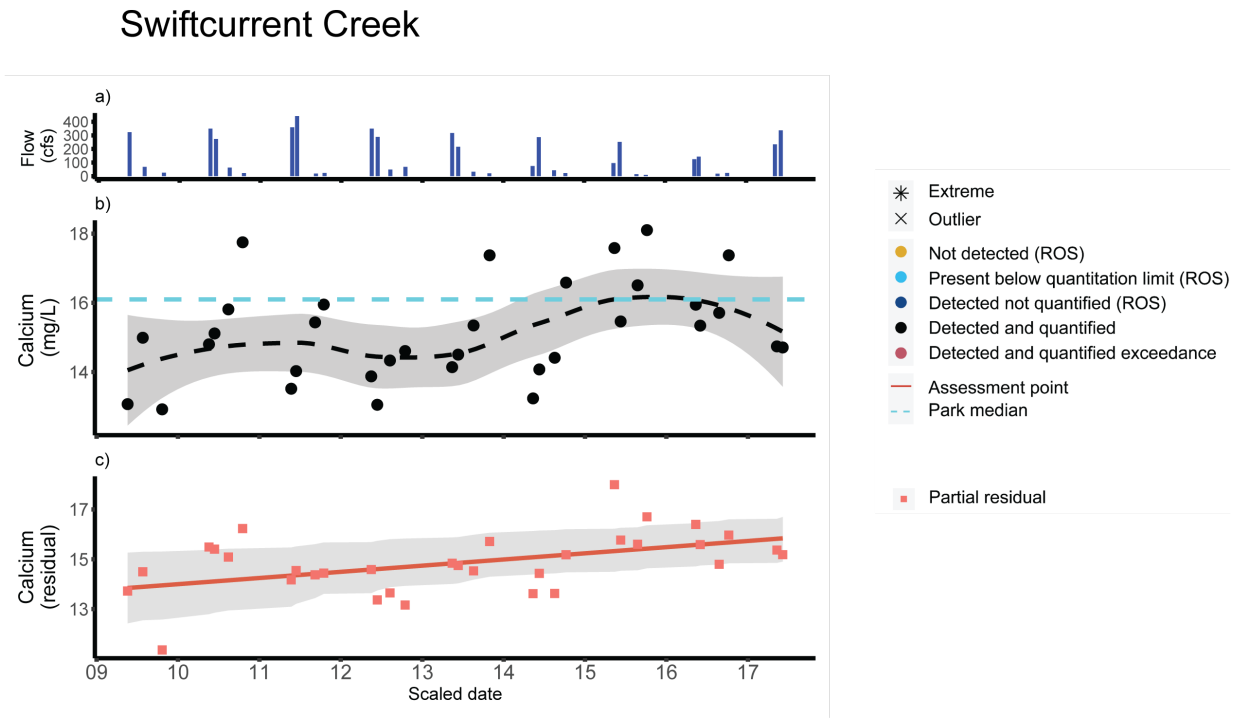


Figure 35. Select trends in calcium (mg/L) in Glacier National Park from 2009 to 2019, Swiftcurrent Creek sentinel stream monitoring site. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). The ecoregional assessment point for calcium is omitted given its off-the-scale value of 1840.7 mg/L, but a 2007–2009 park-scale median is shown (blue dotted line). (c) Partial residuals and the fitted line (significant for this metric) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

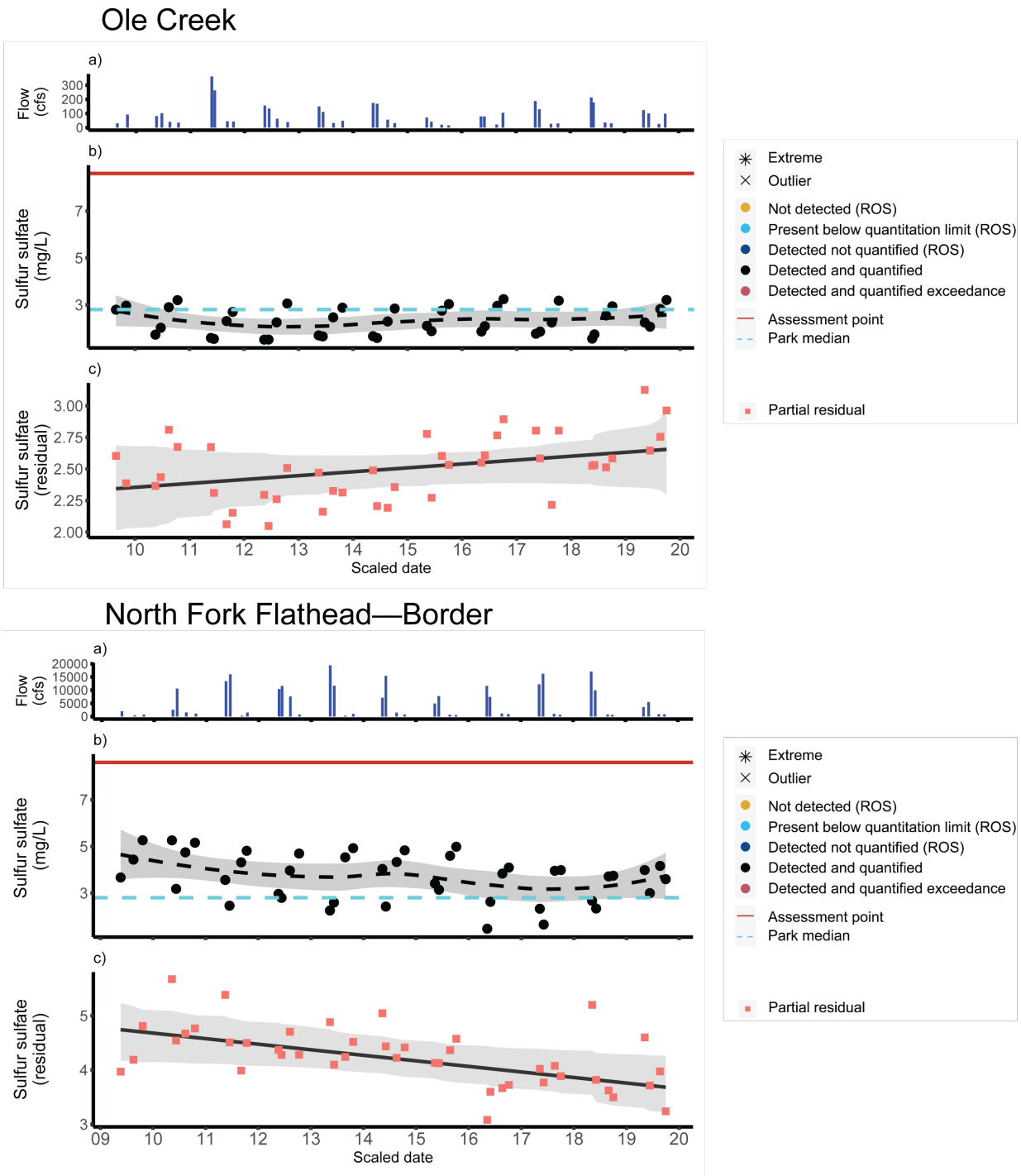


Figure 36. Select trends in sulfate (mg/L as SO4) in Glacier National Park from 2009 to 2019: (top) Ole Creek, (bottom) North Fork Flathead—Border sentinel stream monitoring sites. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment point (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (not significant for this metric) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

Condition

No sites except at the North Fork Flathead—Glacier Rim had values above their ecoregional assessment points. At the North Fork Flathead—Glacier Rim site, results for sulfate were outside the ecoregional assessment points 25% of the time, with some of the highest concentrations in the Glacier National Park dataset. However, these concentrations were still quite low relative to streams outside of Glacier, and with a sample size of only eight over two years, this is not likely meaningful. Otherwise, sulfate, chloride, and calcium median values were typical for large rivers (Table 13).

Discussion and management application

Mast (2013) summarizes patterns in chloride at several reference sites across the United States that indicate concentrations in stream water decreased from the mid-1960s to 2010, likely due to regulation from the Clear Air and Clean Water Acts. Similarly, Sullivan et al. (2018) show that sulfate concentrations generally declined across the United States after these important national policies took hold, particularly in areas where large anthropogenic point sources impact waterbodies. This is often described as one of the successes of US environmental policy, as well as a key demonstration of the importance of long-term monitoring.

Our data continue this flat or negative trend in chloride from 2007 to 2019, except for Swiftcurrent Creek (Figure 36). This site has an undeveloped watershed, but the sample reach itself is embedded in a high use campground, with paved roads and many nearby intensive use (for Glacier) visitor facilities. This development is largely just downstream of where we collect water samples, but a tributary (Wilbur Creek) does drain developed areas and feeds runoff into the main channel of Swiftcurrent Creek upstream of our site. Thus, while chloride concentrations at Swiftcurrent Creek remain low relative to more developed landscapes, the significant increase we observed might be driven by intensifying adjacent land use, road maintenance, and other activities (Figure 37; see the Human Disturbance section above). Future work might isolate and confirm this by sampling above the confluence with Wilbur Creek and/or in Wilbur Creek above and below the potential inputs from the visitor facilities (see Figure 37).



Figure 37. Swiftcurrent Creek sentinel site in Glacier National Park and adjacent land use in 2016. Sample site is indicated by labeled red star. Development on the sample reach's north bank on the right side of this image includes a large campground, housing, visitor services and associated roads and infrastructure. Wilbur Creek, upstream of our sample site, drains much of this developed area.

More recent regional patterns in sulfate deposition and impact to streams may be more complex than the initial successes of the Clean Air and Water Acts brought forth. Borg and Sundbom (2014) show that stream sulfate concentrations can lag deposition rates and suggest that stored soil sulfate may now or could soon be adding to sulfate stream levels. This is possibly because of climate-related changes in runoff routes through the soil profiles following shorter periods of snow cover. Crawford et al. (2019), using data from across the mountainous western United States, including long term sites outside another network park, Rocky Mountain National Park, demonstrate a consistent and widespread pattern of increasing sulfate and base cation concentrations, including calcium specific to mountain watersheds. Sulfide oxidation reaction rates are temperature dependent, with warmer temperatures resulting in higher rates, and these processes occur with a greater magnitude at higher elevations. Kaushal et al. (2021) present a similar and compelling synthesis of data regarding the "salinization syndrome" of a shift in the composition of major ions in aquatic systems (that is not necessarily a high elevation process). This research suggests that recent changes in climate may be stimulating changes to hydrology and/or geomorphic processes, which in turn lead to accelerated weathering of bedrock and elevated levels of ions. The consistency in these patterns across regions and stream types may suggest the possibility of a broad global scale change in stream chemistry, especially in mountain watersheds. While sulfate at the North Fork Flathead—Border sentinel site had a small nonsignificant decrease from 2007 to 2019 (continuing the trend seen in Mast 2007), Reynolds, Ole, and Swiftcurrent Creeks all had small increases in sulfate (Table 13). These sulfate

trends were nonsignificant, but there were positive and significant trends in calcium at Swiftcurrent Creek and positive but nonsignificant trends in calcium and magnesium at all sites. The increases in these cations may be evidence that the climate driven complex story in Crawford et al. (2019) is playing out or beginning to do so in Glacier.

Our data are not conclusive on this matter, and we need additional data to confirm if this is the case in Glacier. Given the demonstrated patterns across the West, and the cascade of effects from these changes, there are broad implications for the chemistry and quality of important surface water resources in the park that will need to be monitored into the future.

3.3.5.2 Alkalinity

Background and relevancy

Alkalinity measures the capacity of water to neutralize acid. Natural levels of alkalinity are important because they protect or buffer against rapid pH changes, which can be harmful to aquatic life. We measure alkalinity as the concentration of calcium carbonate ($CaCO_3$). Sources for these ions are generally natural, including carbonate rocks (i.e., limestone) and reactions in stream catchments when ions in water exchange with those electrostatically bound to a solid, like a clay particle (e.g., the same process as used in household water softeners that removes ions). Some wastewater contains high levels of carbonates from cleaning agents and food residue—possible in the park from localized visitor use of streams but likely of small impact. Prior to effective controls of acid rain in the mid-1970s, alkalinity often varied with levels of sulfuric acid emissions (Likens et al. 1977; Schindler 1988), with increases in alkalinity due to accelerated weathering and dissolution of carbonates from the sulfate emissions (Kaushal et al. 2013) in a complex balance, or lack thereof, with increased levels of acidification from acid rain, reducing alkalinity. More recently, changes in the releases of sulfides (described above), along with evidence for climate driven increases in nitrogen oxides, (i.e., Kuylenstierna et al. 2001), may suggest that aquatic systems are prone to increasing acidification. As noted above in the pH section, this is generally not a good thing for aquatic systems.

Assessment points

There are no relevant regulatory criteria for alkalinity, although the US EPA (EPA 2009) informally recommends a minimum value of 20 mg/L. Levels of 20–200 mg/L are typical of fresh water and alkalinity levels of around 100 mg/L will generally stabilize pH levels in streams, depending on the input of compounds that acidify water. Levels below 10 mg/L indicate that the system is poorly buffered and can be susceptible to changes in pH from natural and/or human-caused sources. While alkalinity is important to stabilize acidity levels, in excess it can also result in a range of effects, including hard water, ammonia toxicity, and algal blooms (Kaushal et al. 2013). We constructed an upper ecoregional assessment point from reference sites in the ecoregions surrounding the park at 144 mg/L and use the EPA's recommended level of 10 mg/ as a lower assessment point. Neither of these have any regulatory authority; rather, they estimate a range of alkalinity in comparable stream sites that are likely in reference condition and provide context for results in Glacier under an expectation of a reference state.

Parkwide 2007–2009 baseline and other previous results

Alkalinity in the 2007–2009 baseline survey was variable and related to watershed size and elevation; smaller and higher elevation sites had lower alkalinity. However, overall, alkalinity levels in Glacier in 2007–2009 did not suggest meaningful disturbance or systemic problems in the park's streams. The population scale median of 61.5 mg/L was well below the ecoregional assessment point and above the lower assessment point. Mast (2007) reported site specific data from 1984 to 2004 with a median of 49 mg/L. Mills et al. (2011) reported a site-specific median value from 2007 to 2008 for the North Fork Flathead—Border of 124 mg/L. Mills et al. (2011) showed small but significant positive trends in alkalinity at specific sites on the North Fork. This trend covers a period of record spanning the late 1970s to the mid-1990s. However, for unclear reasons, a trend in alkalinity was not in Mast (2007), even though the data and methods for these two reports were similar.

Trends

All sites had positive trend coefficients, with the North Fork Flathead—Border site marginally significant (Table 13, Figure 38). This extends the similar trend at this site reported in Mills et al. (2011). Trend models did not indicate any lab effects. Season was significant at all sites (or marginally so), with higher values in the fall and early winter, where flows are much reduced and the relative concentration of the dissolved solutes that elevate alkalinity are likely higher. Flow did play a significant role at two of the four trend sites, but with mixed patterns.

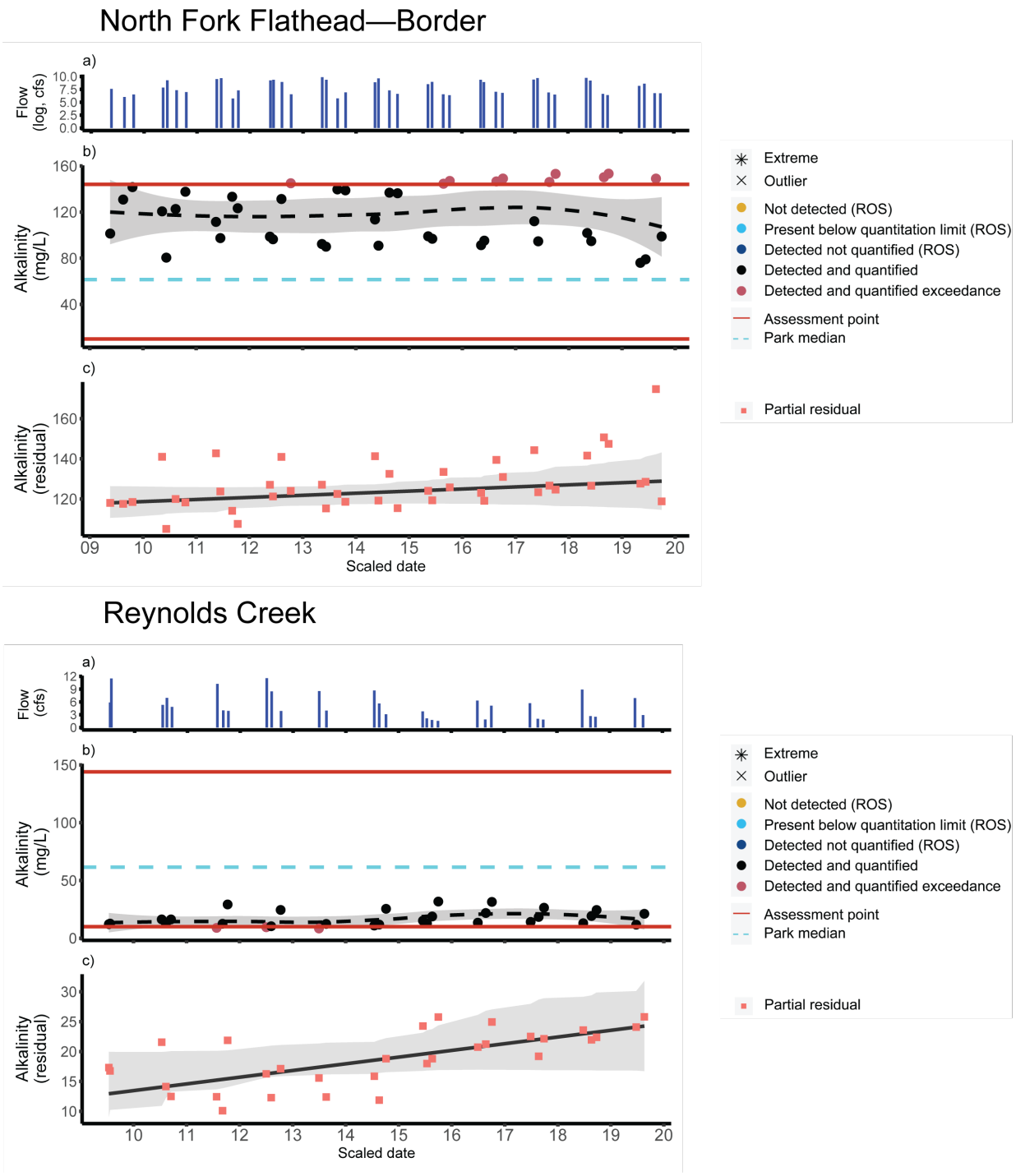


Figure 38. Select trends in alkalinity (mg/L of CaCO3) at sentinel stream monitoring sites in Glacier National Park from 2008 to 2019, (top) North Fork Flathead—Border and (bottom) Reynolds Creek. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment point (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (not significant for this metric) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites.

Condition

Two of the six sites had values outside of the ecoregional assessment points for alkalinity, ranging from 9.4% to 24.4%, and included results on the dilute side or below the lower assessment point (Reynolds Creek) and on the concentrated side (North Fork Flathead—Border; Figure 38) or above the upper assessment point. Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead had median alkalinity values like other large river results and no values exceeded the assessment points (Table 13).

Discussion and management application

Increased stream alkalinization can have environmental implications, including impacts on hardness and salinization of drinking water, alterations of air–water exchange of CO2, and the influence of bicarbonate availability on primary production. Drake et al. (2018) link increased alkalinity to higher air and water temperatures as driven by climate change. As we show above, there have been increases in water temperatures in the park and this is likely to continue. In an extensive survey of eastern streams and rivers in the US, Kaushal et al. (2013) show a strong link to carbonate lithology, with the highest rates of alkalinization in high elevation sites, suggesting these process may be of special concern for sites in Glacier National Park like Reynolds Creek.

Increases in alkalinity on large rivers have been linked to increased anthropogenic land use, especially when coupled with higher levels of precipitation and discharge (i.e., the Mississippi River; Raymond and Cole 2003). However, it is unlikely that alkalinity at our large river sentinel sites on the North and Middle Forks of the Flathead River has or is currently influenced by this; meaningful anthropogenic land use has not occurred in the park nor in watersheds that extend beyond the park, such as in the British Columbia Flathead basin to the park's north or in the wilderness areas of the Bob Marshall to the park's south. However, there can be connections between alkalinity and (increasing) visitation on a more localized scale. In Rocky Mountain National Park, human urine and car exhaust can change nutrient loading, and this can eventually end up in streams and rivers, impacting many aspects of chemistry (i.e., in the Loch Vale watershed, this was between 2% and 15% increase of nitrogen per year over the last decade from visitors in the area; Baron et al. 2021). Finally, much of Glacier is high elevation, where deposition driven increases in acidification can have a large impact (Clow and Campbell 2008). Nitrogen oxide deposition has become more prevalent in more recent years (Ingersoll et al. 2008). Given naturally low levels of alkalinity at most sentinel sites, increased nitrogen oxide deposition may become problematic.

Some of these patterns may move chemistry at our sentinel sites in opposite directions (i.e., acidification vs. increased alkalinity). There are also strong seasonal components, with our data suggesting larger increases in alkalinity in the fall, especially at the North Fork Flathead—Border, and more acidic conditions in the spring (at higher elevations such as Reynolds Creek). All of these warrant further monitoring of alkalinity and associated parameters.

3.3.5.3 Total suspended sediment

Background and relevancy

Total suspended sediment (TSS) is a direct measure of the amount of particulate matter in a stream. TSS is important, influencing other parameters such as nutrient levels and turbidity, and where increasing, excess amounts relative to the stream type and its bedload can lead to degradation of aquatic habitat. Specifically, suspended sediment in salmonid spawning areas in Glacier National Park can cause a reduction in fish embryo survivorship (CSKT and MFWP 2004). TSS has largely natural sources in Glacier, such as soil slump and bank erosion (Mast 2007; Mills et al. 2011). Wildfire in the park and associated management activities can cause often high, but transitory, levels of TSS (Mast and Clow 2008). However, select anthropogenic point disturbances, especially from roads, and complex larger scale indirect interactions among human land use, climate, and hydrology can elevate TSS (Dalby 1983; Appleman et al. 1990; Gildea et al. 2004).

Assessment points

There are no regulatory criteria for TSS. We therefore develop an ecoregional assessment point of 6.7 mg/L derived from TSS values at reference sites from across the ecoregions in which Glacier is embedded (Schweiger et al. 2018). It has no regulatory authority; rather it estimates the upper limits of TSS in comparable stream's sites that are in reference condition and provides context for results in Glacier under an expectation of a reference state. Note that given the importance of sediment in streams, Montana DEQ has created criteria for sediment size and embeddedness (Kusnierz et al. 2013) as presented in our physical habitat methods.

Parkwide 200–2009 baseline and other previous results

The 2007–2009 baseline survey TSS data were variable, with a relationship to watershed size (larger rivers naturally have higher TSS) and stream discharge; however, overall, base flow TSS in Glacier in 2007–2009 did not suggest meaningful disturbance or systemic problems in the park's streams. The population scale median of 0.8 mg/L was well below the ecoregional assessment point. Mast (2007) does not report site-specific data for TSS. Mills et al. (2011) reported a site-specific median base flow value for the North Fork Flathead—Border of 8 mg/L, which is within the range of our data at this site. Mills et al. (2011) showed a small and nonsignificant positive trend in base flow TSS at the North Fork Flathead—Border site. This trend covers a period of record spanning the late 1970s to the mid-1990s. Base flow TSS is often a fraction of what can occur during runoff. During high-flow conditions, sediment loads naturally increase, dramatically reflecting greater kinetic energy for erosion and transport (Hem 1989; Horowitz 2008). For example, sampling during runoff in 2008 on the North Fork Flathead—Border and North Fork Flathead—Glacier Rim had high concentrations (319 mg/L and 308 mg/L, respectively; Mills et al. 2011).

Trends

Trends were mixed, with two streams showing TSS increasing (i.e., Swiftcurrent Creek; Table 13, Figure 39) and two showing TSS decreasing (i.e., North Fork Flathead—Border; Figure 39), but none of these trends were significant. The pattern at the North Fork Flathead—Border site extends results in Mills et al. (2011). Our period of record with TSS was shorter than other parameters, and our estimates of change across time are only from 2014 to 2019. Trend models showed no indication

of any lab effects. Season was significant at two sites, with much higher values in the spring when flows are elevated and export of fine sediments from the watershed into runoff increased. Flow did play a significant role at one site, with increasing flow leading to higher TSS.

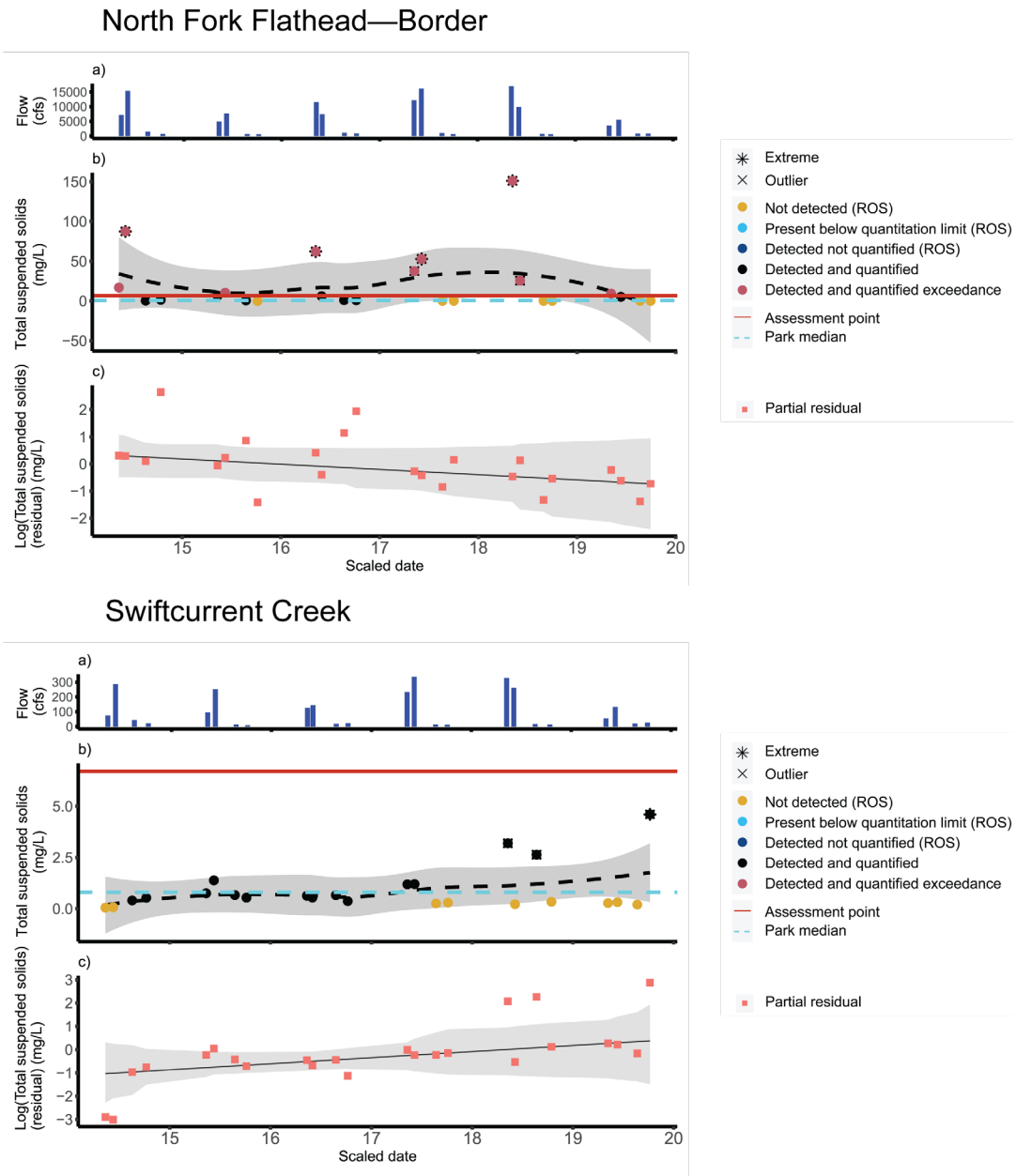


Figure 39. Select trends in total suspended sediment (mg/L) at sentinel stream monitoring sites in Glacier National Park from 2008 to 2019: (top) North Fork Flathead—Border and (bottom) Swiftcurrent Creek. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment point (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (not significant for this metric) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

Condition

Of the six sentinel sites, only North Fork Flathead—Border had TSS values greater than the ecoregional assessment point (33.3%; Figure 39). Most of these were during runoff, as expected, and we do not think this is ecologically meaningful.

Discussion and management application

TSS can be a difficult parameter to collect and analyze given its high variability over small spatial and temporal scales; in fact, TSS can spike during precipitation events. SEI crews follow specific field procedures to optimize the quality of these samples. The challenges are reflected in the high level of censoring (50.5%; Table 13) relative to other parameters. Our methods for imputation should reliably estimate censored values when this rate is below 80%, but this is still worth noting. TSS results should be interpreted with some caution and additional data are needed to better understand sediment dynamics at these sites. The relative percent difference for TSS samples split and analyzed by two different labs was below our acceptance threshold (publication in preparation by Schweiger and others). Lab and field duplicates also had substandard relative percent difference. However, we did not see any contamination in TSS field blanks, and the percentage of outliers and extremes was not excessive.

The processes driving sediment loads in streams and rivers are complex (see Meybeck et al. 2003 for a comprehensive review). Climate change has impacted and will continue to impact precipitation and runoff, as well as vegetative cover in Glacier National Park. These, in part, control soil erosion and sediment transport processes into streams and rivers (Chaplot 2007; Ficklin et al. 2013) and the sediment concentration in streams (e.g., Pruski and Nearing 2002). However, the responses of sediment to changes in climate are nonlinear and seasonal and vary in direction and by region largely due to the interaction between plant production and runoff (Pruski and Nearing 2002). This may explain the inconsistent direction of trend in our TSS data. The variability may also contribute to the lack of significance in our models. The small but positive coefficient suggesting more TSS over time at our alpine site (Reynolds Creek) may be connected (or may become connected) to a loss of snow cover, given shifts of winter precipitation from less erosive snow to more erosive rainfall with increased air temperatures. Decreases in TSS at sites like North Fork Flathead—Border might be related to increases in vegetation biomass driven by climate change-driven temperature and a reduction in erosion with this increased vegetation cover. These explanations will require more research and data to confirm.

3.3.6 Nutrients

We focus on two nutrients in this report, total nitrogen and total phosphorous. We also present an integrated nutrient assessment based on methods from Montana DEQ (Suplee and Sada de Suplee 2016). This requires algae-based productivity (chlorophyll-a, and/or ash free dry mass), a diatom nutrient increaser metric, and a macroinvertebrate nutrient indicator metric. We present more details on diatom and macroinvertebrate metrics in the Bioassessment section below.

3.3.6.1 Total nitrogen and phosphorous

Background and relevancy

Nutrients, primarily nitrogen (N), phosphorous (P), and carbon (C) (i.e., Goldman et al. 1971), are essential for stream ecological function. Total N and P have been shown for decades to limit productivity in streams, whereas carbon is almost never limiting. However, elevated concentrations of nitrogen and/or phosphorus can result in eutrophication, which can cause a variety of problems, including degradation of ecosystem integrity and recreational activities, and impacts to human health (Dubrovsky et al. 2010). Nutrients can be important stressors on aquatic systems. For example, Montana DEQ (2021) found that 35% of river miles and 22% of lake acres surveyed in the state were considered impaired by nutrients. Both natural and anthropogenic drivers affect nutrient concentrations in streams. Natural contributors of nutrients to streams include geology, soil, and groundwater and in situ production. Human activities can accelerate eutrophication via atmospheric deposition, agricultural and urban runoff, leaking septic systems, and sewage discharges (Ellis et al. 2015; EPA 2016b).

Nutrients collected at SEI events in Glacier National Park included several forms of nitrogen (ammonium, nitrite + nitrate, total nitrogen, etc.), phosphorous (total and biologically available orthophosphate), and organic carbon. We include multiple analytes of nutrients in the SEI protocol as they can help elucidate often complex interactions between specific ecological responses that can be tied to a particular form of a nutrient. For example, the native nuisance algae, *Didymosphenia geminate,* may be influenced by ultra-low amounts of organically available phosphorous, as well as total nitrogen, depending on the location (Schweiger et al. 2011; Bothwell et al. 2014). However, following guidance by Montana DEQ (Suplee and Watson 2013), we restrict detailed interpretation in this report to total nitrogen and phosphorous.

Assessment points

Montana DEQ (Suplee et al. 2008; Suplee and Sada de Suplee 2016; Suplee and Watson 2013) has developed nutrient criteria for total N and P and an integrated assessment that are unique and may have relevance for use by Glacier (Table 14). They are specific to summer months (July to September) when stream ecosystems are often most stressed due to low flow conditions and elevated temperature. They vary by ecoregion, with different values in regions where nutrients are naturally elevated versus in regions like around Glacier, where they are naturally lower.

Table 14. Select nutrients at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Percent in exceedance is the percentage of results above (and/or below) the given assessment point(s) (AP). All parameters increase as condition in a stream deteriorates. Values above or equal to the AP are in a non-reference condition. All AP types are regulatory criteria.

Parameter	Site	Trend in partial residual	Trend (p)	Percent in exceedance of AP	AP	Start Date	End Date	Sample size	Percent censored	Number of labs	Number of detection limits	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Nitrogen as N, Total in Water (mg/L)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	0.325	10/4/2007	9/30/2019	8	0.0	2	3	0.100	0.045	0.059	0.032	0.03	0.12
	North Fork Flathead—Border	0.0456	0.179	0.0	0.325	8/20/2007	9/30/2019	46	8.7	3	11	0.100	0.038	0.051	0.037	0.011	0.2
	Swiftcurrent Creek	0.0402	0.568	2.2	0.325	10/18/2007	10/7/2019	45	0.0	3	11	0.100	0.119	0.129	0.067	0.04	0.355
	Reynolds Creek	0.0982	0.993	0.0	0.325	9/11/2008	8/20/2019	32	15.6	3	10	0.100	0.039	0.045	0.023	0.017	0.108
	Ole Creek	0.0323	0.444	0.0	0.325	8/27/2009	10/2/2019	42	14.3	2	9	0.100	0.051	0.058	0.034	0.015	0.136
	Middle Fork Flathead	–[a]	–[a]	0.0	0.325	8/28/2018	9/30/2019	6	0.0	1	1	0.100	0.075	0.088	0.038	0.06	0.16
Phosphorus as P, Total in Water (mg/L)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	0.025	10/4/2007	9/30/2019	8	50.0	2	3	0.010	0.002	0.002	0.002	0	0.005
	North Fork Flathead—Border	−0.0113	0.357	15.2	0.025	8/20/2007	9/30/2019	46	21.7	3	10	0.010	0.005	0.015	0.028	0	0.138
	Swiftcurrent Creek	−0.0894	0.981	0.0	0.025	10/18/2007	10/7/2019	44	34.1	3	10	0.010	0.002	0.003	0.003	0	0.01
	Reynolds Creek	0.1303	0.567	0.0	0.025	9/11/2008	8/20/2019	32	34.4	3	9	0.010	0.001	0.003	0.003	0	0.013
	Ole Creek	−0.0880	0.963	0.0	0.025	8/27/2009	10/2/2019	42	45.2	2	8	0.010	0.001	0.004	0.005	0.001	0.019

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend models for chlorophyll-a/ash free dry mass do not include seasonal, lab, or flow terms.

Table 14 (continued). Select nutrients at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Percent in exceedance is the percentage of results above (and/or below) the given assessment point(s) (AP). All parameters increase as condition in a stream deteriorates. Values above or equal to the AP are in a non-reference condition. All AP types are regulatory criteria.

Parameter	Site	Trend in partial residual	Trend (p)	Percent in exceedance of AP	AP	Start Date	End Date	Sample size	Percent censored	Number of labs	Number of detection limits	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Phosphorus as P, Total in Water (mg/L) (continued)	Middle Fork Flathead	–[a]	–[a]	0.0	0.025	8/28/2018	9/30/2019	6	100.0	1	1	0.010	0.001	0.001	0.000	0.001	0.001
Ash free dry mass in periphyton (mg/m^2)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	35	8/25/2007	8/20/2019	4	0	1	1	6.800	13.269	14.738	11.836	1.972	30.44
	North Fork Flathead—Border	0.7674[b]	0.116[b]	0.0	35	8/20/2007	8/22/2019	11	0	1	1	6.800	8.648	8.636	4.484	2.727	18.264
	Swiftcurrent Creek	0.4311[b]	0.490[b]	0.0	35	10/18/2007	8/22/2019	13	0	1	1	6.800	10.592	10.436	7.875	3.181	33.298
	Reynolds Creek	−0.0853[b]	0.838[b]	0.0	35	9/11/2008	8/20/2019	12	0	1	1	6.800	6.473	7.430	3.135	3.265	12.839
	Ole Creek	0.6087[b]	0.399[b]	0.0	35	8/27/2009	8/21/2019	11	0	1	1	6.800	5.964	6.392	3.892	0.994	13.17
	Middle Fork Flathead	–[a]	–[a]	0.0	35	8/28/2018	8/19/2019	2	0	1	1	6.800	23.633	23.633	9.702	16.773	30.494
Chlorophyll a in periphyton (mg/m^2)	North Fork Flathead—Glacier Rim	–[a]	–[a]	0.0	125	8/25/2007	8/20/2019	4	0	1	1	5.900	0.785	1.418	1.398	0.597	3.503
	North Fork Flathead—Border	−0.1412[b]	0.782[b]	0.0	125	8/20/2007	8/22/2019	11	0	1	1	5.900	4.963	4.927	3.593	0.385	12.274

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend models for chlorophyll-a/ash free dry mass do not include seasonal, lab, or flow terms.

Table 14 (continued). Select nutrients at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Percent in exceedance is the percentage of results above (and/or below) the given assessment point(s) (AP). All parameters increase as condition in a stream deteriorates. Values above or equal to the AP are in a non-reference condition. All AP types are regulatory criteria.

Parameter	Site	Trend in partial residual	Trend (p)	Percent in exceedance of AP	AP	Start Date	End Date	Sample size	Percent censored	Number of labs	Number of detection limits	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Chlorophyll a in periphyton (mg/m^2) (continued)	Swiftcurrent Creek	−0.3727 [b]	0.718 [b]	0.0	125	10/18/2007	8/22/2019	13	0	1	1	5.900	4.380	6.625	6.377	0.053	18.733
	Reynolds Creek	−0.4760 [b]	0.304 [b]	0.0	125	9/11/2008	8/20/2019	12	0	1	1	5.900	2.538	4.010	5.094	0.093	18.461
	Ole Creek	−0.4010 [b]	0.365 [b]	0.0	125	8/27/2009	8/21/2019	11	0	1	1	5.900	1.338	2.840	3.309	0.04	10.218
	Middle Fork Flathead	– [a]	– [a]	0.0	125	8/28/2018	8/19/2019	2	0	1	1	5.900	0.285	0.285	0.066	0.239	0.332

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend models for chlorophyll-a/ash free dry mass do not include seasonal, lab, or flow terms.

Parkwide 2007–2009 baseline and other previous results

In general, concentrations of total nitrogen and phosphorous in the 2007–2009 baseline survey were low and near or below their detection limits, and neither total nitrogen nor phosphorous concentrations suggested systemic nutrient problems in the park's streams. The population scale medians of 0.15 and 0.01 mg/L were well below the Montana DEQ criteria of 0.325 mg/L and 0.025 mg/L for total nitrogen and total phosphorous, respectively. Select sites near Lake McDonald inholdings and some sites located near roads, especially on the park's boundary, had some elevated values. However, bioassessment results presented in Schweiger et al. (2018) suggested it was unlikely that these potentially anthropogenic driven nutrient levels had meaningful impacts on the park's streams and rivers.

Mast (2007) reported site-specific data from 1984 to 2004 with a median of 0.0038 mg/L for total phosphorous. This is considerably lower than our 2007–2009 park-scaled median value of 0.01 mg/L. This is likely due to sample design issues in Mast (2007), but without further work, this is not resolvable. Mills et al. (2011) reported a site-specific median value from 2007 to 2008 for the North Fork Flathead—Border of 0.01 mg/L, coincidentally identical to our park-scaled value. Mills et al. (2011) also report total nitrogen at 0.06 mg/L, again lower than the park scaled median value from Schweiger et al. (2018). Mills et al. (2011) showed an insignificant positive trend in total phosphorous at the North Fork Flathead—Border site. This trend covers a period of record spanning the late 1970s to 2008. However, in Mast (2007), the trend at this same site was significant and negative but over a shorter period of 1974 to 1993.

Trends

While trend coefficients were mixed (Table 14), no models suggested any trend was significant. There was no indication of any lab effects in any model. Season was significant at Ole Creek and marginally so at North Fork Flathead—Border, with higher values in the spring when flows are elevated and runoff contributes more nutrients to streams and rivers. Flow did play a significant role in two of the four models for total phosphorous, with higher concentration at greater flow. Note that total N and total P do have a unique a priori model form, with a log transformation of the response following Schertz et al. (1991) given a strong connection to sediment dynamics influenced by flow. Chlorophyll-a/ash free dry mass data also have a different structure, with base flow samples only, no lab term, and no seasonal component.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites.

In general, total nitrogen and total phosphorous results likely can be interpreted without concern for QAQC issues. Given the naturally low concentrations of total nitrogen and total phosphorous in the park's waters, there was a higher level of censoring in these results and several data points were successfully estimated by imputation. QAQC results (in preparation by Schweiger and others) also indicate that total nitrogen and total phosphorous samples split and analyzed by two different labs were not below the acceptance threshold. Lab and field duplicates had acceptable relative percent differences and the percentage of outliers and extremes was not excessive. There was some

contamination in select field blanks, but any paired sample with contamination was removed from all analyses.

Condition

There was a single exceedance of the Montana DEQ criteria we use as an assessment point for total nitrogen at Swiftcurrent Creek in the spring of 2010 (Figure 40). This is likely due to naturally elevated levels of nitrogen that can occur during spring runoff. Concentrations of total phosphorous at North Fork Flathead—Border were above the assessment point of 0.025 mg/L in 15.2% of samples Figure 41. Nearly all of these also occurred during spring runoff. Otherwise, total phosphorous concentrations at this site were very low (several values were below detection and estimated with Regression on Order Statistics (ROS)). We explore these results in more detail below using the Montana DEQ integrated nutrient assessment protocol.

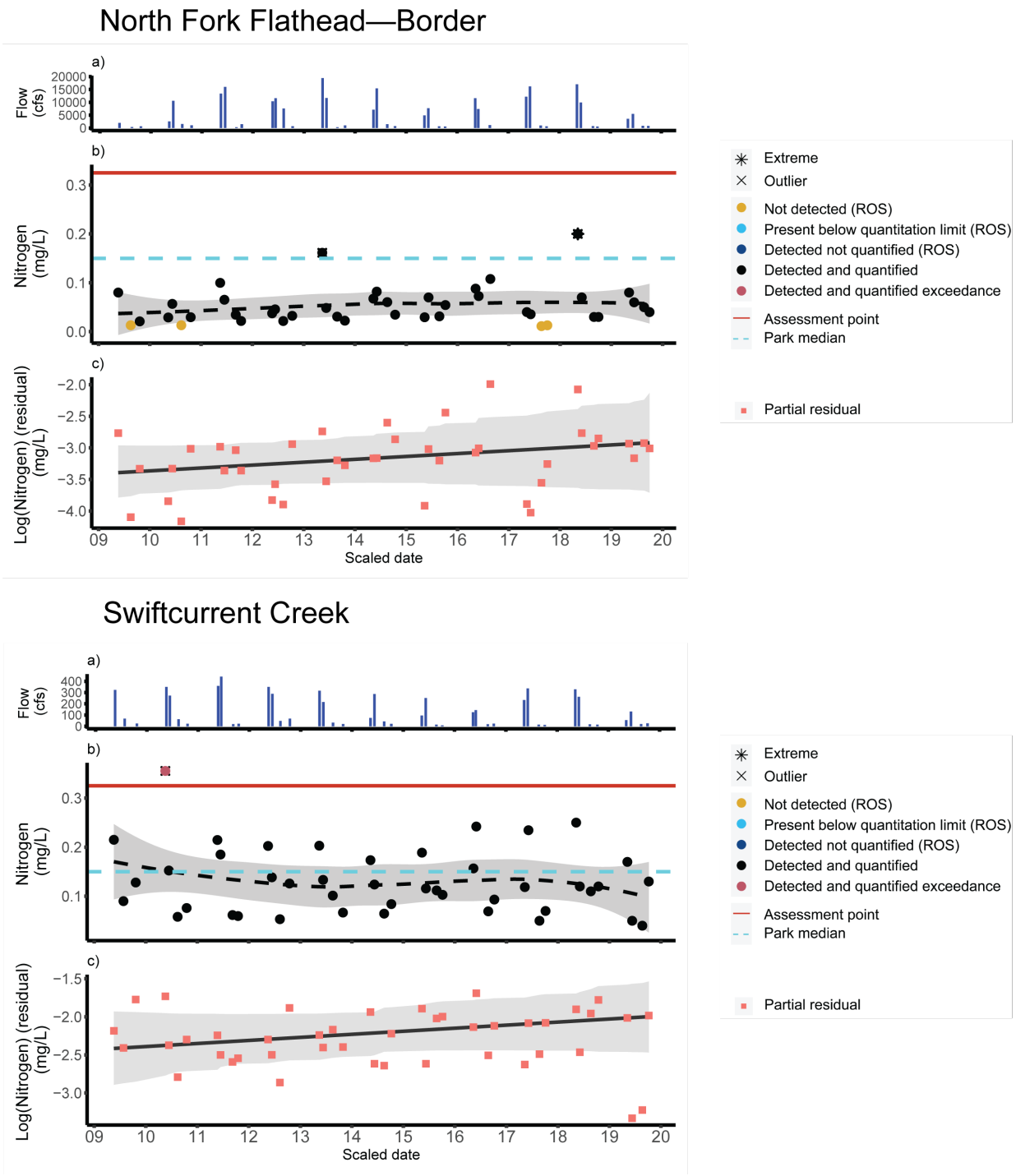


Figure 40. Select trends in total nitrogen (in water, mg/L) in Glacier National Park sentinel stream monitoring from 2009 to 2019: (top) North Fork Flathead—Border sentinel site and (bottom) Swiftcurrent Creek. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment point (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (not significant for this metric) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

Median total nitrogen and total phosphorous values at sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead were like other large river sentinel sites and no values exceeded an assessment point (Table 14, Figure 41). Total nitrogen at the Middle Fork Flathead site may have been slightly elevated, but more data are needed to confirm this.

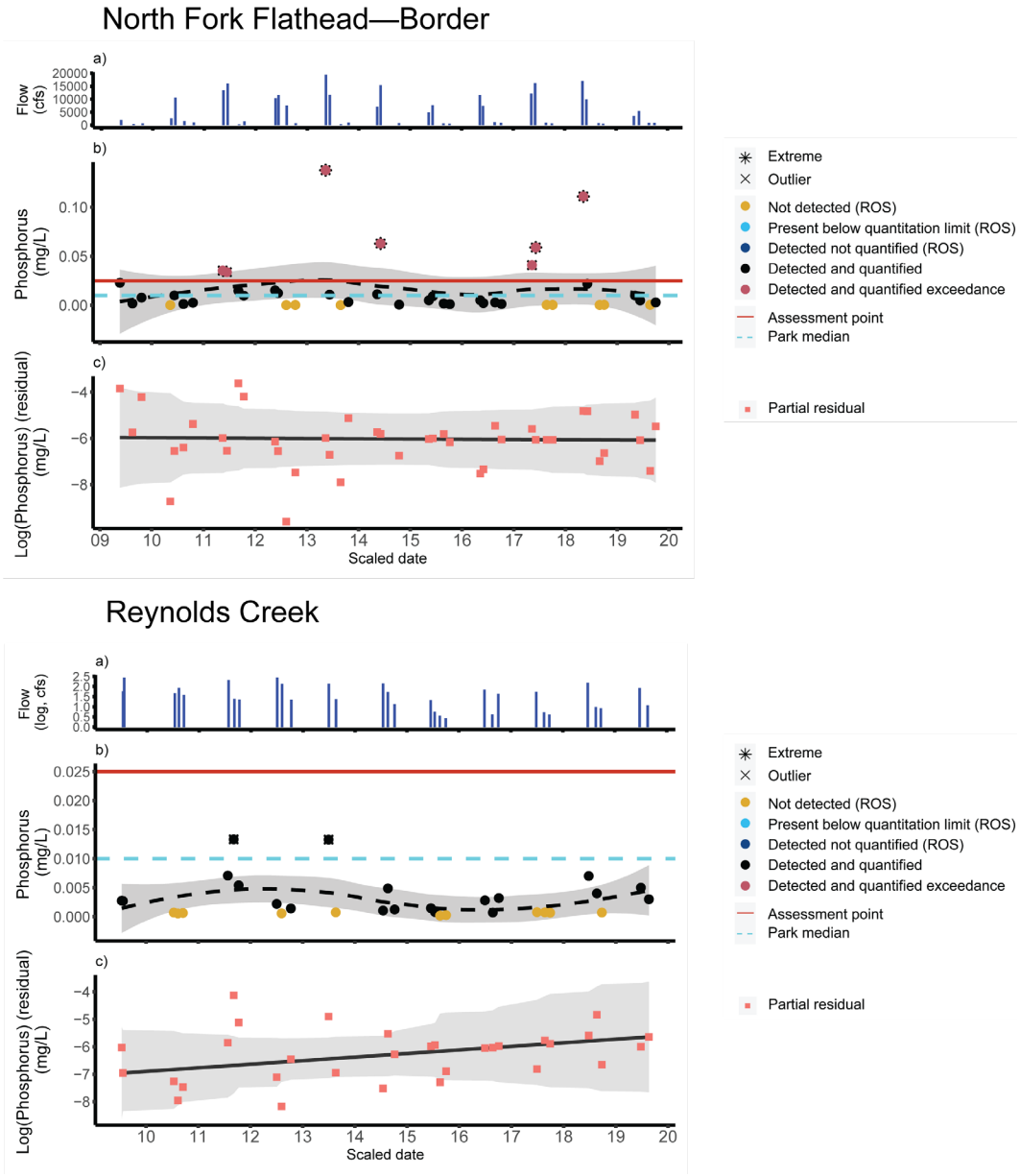


Figure 41. Select trends in total phosphorous in Glacier National Park sentinel stream monitoring sites from 2009 to 2019: (top) North Fork Flathead—Border and (bottom) Reynolds Creek. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment point (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (not significant for this metric) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

Discussion and management application

Overall, we suspect that most elevated nutrient concentrations found at our sentinel sites are largely, if not entirely, natural in their source(s). There are several focused research efforts in and near the park that support this. For example, Stanford et al. (1994) demonstrated that dissolved nitrate can concentrate in the center of undisturbed alluvial aquifers in the park, with 12%–17% of the base flow nitrate load of the river downstream from the aquifer derived from nitrification within the aquifer. They also showed that organically available phosphorus can be up to an order of magnitude higher in floodplain groundwater. High solute loads in groundwater can discharge into surface waters, producing intense zonation in nutrients (revealed by localized mats of green algae, such as *Homoeothrix* and/or *Calothrix*; Figure 42). Schweiger et al. (2018) presents a model of total phosphorous in the park using our 2007–2009 survey data that suggests natural variables like sediment concentrations and not general human disturbance (HDI as presented above) are best related to total phosphorous concentrations.



Figure 42. Algal bloom, (probably dominated by the cyanobacteria genera Homoeothrix and/or Calothrix) on a backwater of the North Fork of the Flathead River in British Columbia, Canada, just north of Glacier National Park, 2008. The bloom was likely caused by groundwater upwelling from deep floodplain aquifers. (NPS / BILLY SCHWEIGER).

Climate change likely plays an important role in nutrient dynamics (Jeppesen et al. 2010). Increasing nutrient levels are often driven by structural connections to increasing temperatures and altered flow regimes and subsequent changes to production. Warmer water can better assimilate increasing nutrients while declining temperatures reduce solubility and may result in direct precipitation of those solutes. This variability may be reflected in the mixed patterns in total nitrogen and total phosphorous at our sites. Using randomized design survey data from across the US from 2000 to 2014 (Stoddard et al. 2016) suggested total phosphorous increased across regional scales, especially in relatively undisturbed watersheds like Glacier National Park. Unlike total phosphorous, however, total nitrogen does not appear to be increasing in undisturbed catchments. Stoddard et al. (2016) conclude that increasing concentrations in undeveloped catchments cannot be attributed to either point or common non-point sources of total phosphorous. Rather the mechanisms are likely a combination of recovery from atmospheric acidification and a subsequent increase in leaching of nutrients from catchment soils, elevated natural movement of phosphorus via increased or changes in timing of runoff, or an increased atmospheric contribution of phosphorus to streams. Yet phosphorus deposition has recently been identified as a potentially important source to alpine streams in Europe and North America (Brahney et al. 2014), with ultimate sources from increased fossil fuel combustion and forest fires.

Schweiger et al. (2018) show that streams in the park were phosphorous limited in 2007–2009. This may change if phosphorous concentrations increase in the park, and this has important consequences to potential eutrophication (that is tightly linked to phosphorous). In general, our current results and the body of research suggesting phosphorous concerns warrant ongoing monitoring of nutrients in the park. In 2020, Glacier began its own monitoring of dry deposition of phosphorous at Apgar, also in response to these concerns.

3.3.6.2 Integrated nutrient assessment

Background and relevancy

Montana DEQ provides a detailed protocol for comprehensive evaluation of possible nutrient issues (Suplee and Sada de Suplee 2016). The method uses a tiered approach based on nutrient concentrations (total N and P) plus a series of support criteria, including algae-based productivity (chlorophyll-a, ash free dry mass), a diatom nutrient increaser metric, and a macroinvertebrate nutrient indicator metric. Suplee and Sada de Suplee (2016) present a decision support system that guides evaluation of these four data elements. Montana DEQ defines statistical tests, associated parameters for evaluating results, and a hierarchical decision process to arrive at a final assessment of a site's nutrient status.

The network SEI protocol provides data suitable for use in the Montana DEQ method, with a few limitations. To establish a regulatory exceedance, the Montana DEQ method requires minimum water chemistry grab sample sizes over a growing season of 12 for total nitrogen or phosphorous, three for chlorophyll-a/ash free dry mass samples, and two for the diatom increaser and macroinvertebrate nutrient metrics. The SEI method only includes four chemistry samples, one chlorophyll-a,/ash free dry mass, and one diatom/macroinvertebrate sample during most growing seasons.

As an explicit regulatory objective is not part of our monitoring protocol, this smaller sample size may be acceptable as long as the consequences on statistical power are taken into consideration. Should a network sentinel site ever appear at risk of a serious nutrient exceedance, with a clear connection to a human cause (see Suplee et al. (2005) or Kusnierz et al. (2013)), we can adjust sampling intensity accordingly.

Assessment points

Montana DEQ (2019) defines regulatory criteria that we adopt as assessment points for each of the four components of the integrated nutrient assessment. These are presented and evaluated in relevant sections throughout this report. NPS uses these as assessment points in our work, but any regulatory authority related to them rests with Montana DEQ and its interaction with Glacier National Park resource management.

Parkwide 2007–2009 baseline and other previous results

Only a single site-year failed the integrated nutrient assessment in the baseline 2007–2009 survey: Rubideau Creek in 2008, located downstream of the park's water supply system. Given the proximity of the site to a utility with known (but now fixed) issues (NPS 2002), excessive nutrient levels in Rubideau Creek were informally classified as having a potential human cause.

Trends

There were no significant or marginally significant trends in the integrated nutrient assessment at any sentinel site over our period of record. The final structure for these results (in its simplest form) is a "yes" or "no" regarding nutrient impairment. As all results in our data were "no," there is no real way to apply our current trend models to this unimodal response.

Condition

Table 14 presents the sites in our data that had any exceedance of the total N and total P water chemistry criteria (as simple percentages). The Montana DEQ integrated nutrient assessment methods specify statistical tests of these data (T tests and binomial tests) that compare mean values of total N or P to the Montana DEQ criteria or observed versus expected rates of exceedances. In summary, no years at any sentinel site had concentrations of these nutrients significantly above the assessment point. Likewise, no sites had any exceedances in chlorophyll-a/ash free dry mass samples or the diatom increaser metric (section 3.5.5 below). However, several sites had macroinvertebrate nutrient indicator metric values above the assessment point (see section 3.5.3 below).

The chlorophyll-a/ash free dry mass samples, diatom, and macroinvertebrate bioassessment data are secondary criteria and they cannot in isolation establish a nutrient impairment. Sites can have a nutrient impairment if all three supporting biocriteria were in exceedance, but water chemistry results are not, however; this never happened in at our sites over our period of record.

Discussion and management applications

Importantly, we lack the recommended sample sizes to fully implement the Montana DEQ method. With only four replicates of water chemistry per growing season, the power of the T test and exact binomial tests as specified by Montana DEQ is low and our overall final interpretations should be considered exploratory. With these caveats in mind, the integrated nutrient assessment results for our

sentinel sites suggest there were no nutrient impairments over our period of record. Where we occasionally see elevated values, such as total phosphorous levels coupled with high values of the macroinvertebrate nutrient metric but not the diatom increaser or productivity metrics, it is possible that sampling timing may have missed algal peak and/or there may be possible false negative or results close to the decision threshold. This describes the scenario in some years at the North Fork Flathead—Border site and suggests ongoing monitoring and evaluation is warranted.

3.3.7 Trace elements (metals) in sediment

We focus on five trace elements (hereafter metals) in sediment: arsenic, cadmium, copper, zinc, and total mercury.

3.3.7.1 Arsenic, cadmium, copper, zinc, and mercury in sediment

Background and relevancy

Metals in sediments are of greater ecological relevancy than when in water, given longer residence time in streams and rivers. Water moves into and out of a stream rapidly. This can cause wide variations in concentrations over short time periods. Sediment moves more slowly, and contamination in sediment may therefore have longer lasting and greater effects on aquatic life (Hem 1992) and can have serious human health impacts (Morais et al. 2012). Mercury is especially toxic to aquatic life and can cause serious drinking water issues. However, given the cost and complexity of directly sampling bioavailable methylmercury, we instead analyze samples for total mercury. These two forms are correlated (Rolfhus et al. 2003), as total mercury is converted by microorganisms into methylmercury (we assume a linear positive correlation between total and methylmercury).

Arsenic, cadmium, copper, zinc, and mercury occur naturally in most soils and bedrock in and around Glacier National Park. Natural emissions of these metals can occur via forest fires and rock weathering. There are limited examples of known human sources of metals in and around Glacier. Perhaps the most well-known are the East Kootenay coalfields that underlie the Canadian headwaters of the North Fork of the Flathead River. This has been an area of active mineral exploration over the years, with several coal and coalbed methane developments proposed during the early 2000s (Hauer et al. 2007; Figure 43). Coal and coalbed methane development in these basins would negatively affect water quality of the North Fork by increasing concentrations of suspended sediment, nutrients, and metals in the river (Kootenay Air and Water Quality Study 1978; International Joint Commission 1988; Dalby 1983; Appleman et al. 1990). In 2005, as a response to proposed energy development, an international meeting of scientists and resource managers was convened at the Crown of the Continent Synthesis Workshop. Participants at the workshop identified baseline water-quality and biological data for the North Fork as a critical information gap (Hauer and Sexton 2010). The Province of British Columbia issued a Memorandum of Understanding banning energy resource development activity in the North Fork Basin in 2010 (Province of British Columbia 2010; American Rivers 2010). Other possible anthropogenic sources of these metals include vehicle exhaust (aircraft still release lead; EPA 2022); smelting (arsenic, copper, and zinc); insecticides (arsenic); and large scale burning of fossil fuels across the region and globe (nickel, vanadium, mercury, selenium and tin; Masindi and Muedi 2018).



Figure 43. Active mineral exploration just north of Glacier National Park. (Left) Coal bed methane site in the British Columbia Flathead River watershed, Canada, just north of Glacier National Park, 2008; (right) ground water exploration well near the North Fork of the Flathead River in British Columbia, Canada, with possible evidence of metal leachate. (NPS / BILLY SCHWEIGER).

Assessment points

Assessment points for metals in sediment are hard to come by. The State of Montana has designated sediment quality guidelines (i.e., not regulatory criteria) following MacDonald et al. (2000). These are based on researched that developed a unifying synthesis across multiple published studies of 28 chemical substances and the broad spectrum of sediment quality guidelines for these in aquatic systems. Two classes of criteria were developed: Threshold Effects Concentrations (TEC) "below which adverse effects are not expected"—we interpret this as a reference condition threshold—and a Probable Effects Concentrations (PEC), "above which adverse effects are expected to occur more often than not"—which we interpret as a non-reference threshold.

Both TECs and PECs are likely more relevant in watersheds with greater levels of anthropogenically sourced metals. For example, we use these as assessment points in another network park, Grant-Kohrs Ranch National Historic Site, which is an active Superfund site due to metals. We informally apply these to our Glacier National Park sentinel sites, but caution that because they were developed for systems where metal contamination is expected, they are likely conservative or define thresholds at a higher concentration than where true metal-based issues might occur in a wilderness park like Glacier.

Trends

The direction and significance of trends in sediment metals were mixed (Table 15, Figure 44, Figure 45); however, of the 15 models in this group, five were significant, four with positive trend coefficients. An additional two were nearly marginally significant, both with a positive coefficient. Specific key trends include significantly increasing copper and zinc at Swiftcurrent Creek, cadmium at Reynolds Creek, and arsenic at North Fork Flathead—Border. In contrast, arsenic at Ole Creek significantly decreased. For most site parameters, only one lab was used. For those with at least two labs, only cadmium at North Fork Flathead—Border had a significant lab effect, and this was effectively dealt with in our quantile regression model. Season was marginally significant at North Fork Flathead—Border for zinc, with higher values in the spring when flows are elevated. However, flow was not significant in any model. Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites.

Table 15. Select trace elements (metals) in sediment at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any percent in exceedance value above or equal to 25% is in bold. All AP types are ecological Threshold for Effects Concentrations (TEC). All parameters increase as condition in a stream deteriorates. Values above or equal to the AP are in a non-reference condition.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside of AP	AP	Start date	End date	Sample size	Percent censored	Number of labs	Number of detection limits	Median	Mean	Standard deviation	Min	Max
Arsenic, total in sediment (mg/kg)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	9.79	8/29/2018	9/30/2019	6	0.0	1	5	2.500	2.450	0.418	1.9	3.1
	North Fork Flathead—Border	0.3442	**0.108** [b]	5.4	9.79	5/21/2009	9/30/2019	37	5.4	2	15	5.900	6.351	4.037	0.816	25.2
	Swiftcurrent Creek	−0.0684	0.727	9.1	9.79	5/20/2009	10/6/2015	22	4.5	1	8	7.450	7.272	1.595	2.184	9.8
	Reynolds Creek	−0.4814	0.285	**55.6** [c]	9.79	7/15/2009	10/1/2015	18	11.1	1	7	10.100	9.376	3.090	1.176	14.5
	Ole Creek	−0.2218	**0.010** [b]	0.0	9.79	8/27/2009	10/5/2015	20	0.0	1	6	3.450	3.535	0.612	2.7	4.7
	Middle Fork Flathead	– [a]	– [a]	0.0	9.79	8/28/2018	9/30/2019	6	0.0	1	6	2.500	2.883	0.956	2	4.7
Cadmium, total in sediment (mg/kg)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	0.99	8/29/2018	9/30/2019	6	0.0	1	6	0.053	0.057	0.010	0.049	0.076
	North Fork Flathead—Border	0.0204	0.170	0.0	0.99	5/12/2010	9/30/2019	34	5.9	2	16	0.235	0.268	0.118	0.002	0.4
	Swiftcurrent Creek	N/A [d]	N/A [d]	0.0	0.99	5/20/2010	10/6/2015	19	89.5	1	6	0.007	0.028	0.069	0.001	0.3

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend (p) is significant ($p \leq 0.10$)

[c] Percent outside value is ≥ 25%.

[d] N/A Larger number of censored values that could not be estimated with ROS prohibit trend models for these parameters.

Table 15 (continued). Select trace elements (metals) in sediment at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any percent in exceedance value above or equal to 25% is in bold. All AP types are ecological Threshold for Effects Concentrations (TEC). All parameters increase as condition in a stream deteriorates. Values above or equal to the AP are in a non-reference condition.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside of AP	AP	Start date	End date	Sample size	Percent censored	Number of labs	Number of detection limits	Median	Mean	Standard deviation	Min	Max
Cadmium, total in sediment (mg/kg) (continued)	Reynolds Creek	0.0611	**0.022** [b]	0.0	0.99	7/14/2010	10/1/2015	16	25.0	1	5	0.200	0.207	0.120	0.009	0.4
	Ole Creek	N/A [d]	N/A [d]	0.0	0.99	5/18/2010	10/5/2015	19	94.7	1	6	0.028	0.028	0.026	0.003	0.1
	Middle Fork Flathead	– [a]	– [a]	0.0	0.99	8/28/2018	9/30/2019	6	0.0	1	6	0.041	0.041	0.012	0.027	0.057
Copper, total in sediment (mg/kg)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	31.6	8/29/2018	9/30/2019	6	0.0	1	5	5.550	6.583	2.845	4.1	12
	North Fork Flathead—Border	0.0246	0.943	0.0	31.6	5/21/2009	9/30/2019	37	0.0	2	13	7.900	8.303	2.236	4.1	13.1
	Swiftcurrent Creek	0.8942	**0.050** [b]	0.0	31.6	5/20/2009	10/6/2015	22	0.0	1	3	11.600	12.100	3.853	7	26
	Reynolds Creek	0.0395	0.923	0.0	31.6	7/15/2009	10/1/2015	18	0.0	1	3	14.000	14.144	1.579	11.5	18.5
	Ole Creek	−0.0925	0.450	0.0	31.6	8/27/2009	10/5/2015	21	0.0	1	3	8.400	8.514	0.644	7.7	10
	Middle Fork Flathead	– [a]	– [a]	0.0	31.6	8/28/2018	9/30/2019	6	0.0	1	6	9.050	8.850	1.826	6	11

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend (p) is significant ($p \leq 0.10$)

[c] Percent outside value is ≥ 25%.

[d] N/A Larger number of censored values that could not be estimated with ROS prohibit trend models for these parameters.

Table 15 (continued). Select trace elements (metals) in sediment at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any percent in exceedance value above or equal to 25% is in bold. All AP types are ecological Threshold for Effects Concentrations (TEC). All parameters increase as condition in a stream deteriorates. Values above or equal to the AP are in a non-reference condition.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside of AP	AP	Start date	End date	Sample size	Percent censored	Number of labs	Number of detection limits	Median	Mean	Standard deviation	Min	Max
Mercury, total in sediment (mg/kg)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	0.18	8/29/2018	9/30/2019	6	50.0	1	3	0.006	0.007	0.007	0.001	0.016
	North Fork Flathead—Border	−0.0001	0.636	0.0	0.18	5/21/2009	9/30/2019	29	48.3	2	16	0.007	0.012	0.012	0.001	0.044
	Swiftcurrent Creek	N/A [d]	N/A [d]	0.0	0.18	5/20/2009	10/16/2012	14	100.0	1	11	0.004	0.005	0.002	0.001	0.01
	Reynolds Creek	N/A [d]	N/A [d]	0.0	0.18	7/15/2009	10/10/2012	11	90.9	1	8	0.005	0.007	0.007	0.002	0.025
	Ole Creek	N/A [d]	N/A [d]	0.0	0.18	8/27/2009	10/17/2012	13	92.3	1	8	0.009	0.014	0.011	0.003	0.044
	Middle Fork Flathead	– [a]	– [a]	0.0	0.18	8/28/2018	9/30/2019	6	50.0	1	6	0.005	0.006	0.005	0.001	0.013
Zinc, total in sediment (mg/kg)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	121	8/29/2018	9/30/2019	6	0.0	1	6	27.000	29.333	7.711	22	42
	North Fork Flathead—Border	0.3775	0.690	0.0	121	5/21/2009	9/30/2019	37	0.0	2	19	35.800	35.927	7.545	19	48.2
	Swiftcurrent Creek	1.2776	**0.044** [b]	0.0	121	5/20/2009	10/6/2015	22	0.0	1	13	31.350	30.964	4.655	20.2	40.2

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend (p) is significant ($p \leq 0.10$)

[c] Percent outside value is ≥ 25%.

[d] N/A Larger number of censored values that could not be estimated with ROS prohibit trend models for these parameters.

Table 15 (continued). Select trace elements (metals) in sediment at stream ecological integrity sentinel sites monitored in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any percent in exceedance value above or equal to 25% is in bold. All AP types are ecological Threshold for Effects Concentrations (TEC). All parameters increase as condition in a stream deteriorates. Values above or equal to the AP are in a non-reference condition.

Parameter	Site	Trend in partial residual	Trend (p)	Percent outside of AP	AP	Start date	End date	Sample size	Percent censored	Number of labs	Number of detection limits	Median	Mean	Standard deviation	Min	Max
Zinc, total in sediment (mg/kg) (continued)	Reynolds Creek	−0.1540	0.827	0.0	121	7/15/2009	10/1/2015	18	0.0	1	9	48.750	48.344	2.867	42.2	53.3
	Ole Creek	0.7060	0.143	0.0	121	8/27/2009	10/5/2015	21	0.0	1	11	31.500	32.205	2.691	27.3	36.6
	Middle Fork Flathead	– [a]	– [a]	0.0	121	8/28/2018	9/30/2019	6	0.0	1	6	24.500	25.333	4.633	20	31

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend (p) is significant ($p \leq 0.10$)

[c] Percent outside value is ≥ 25%.

[d] N/A Larger number of censored values that could not be estimated with ROS prohibit trend models for these parameters.

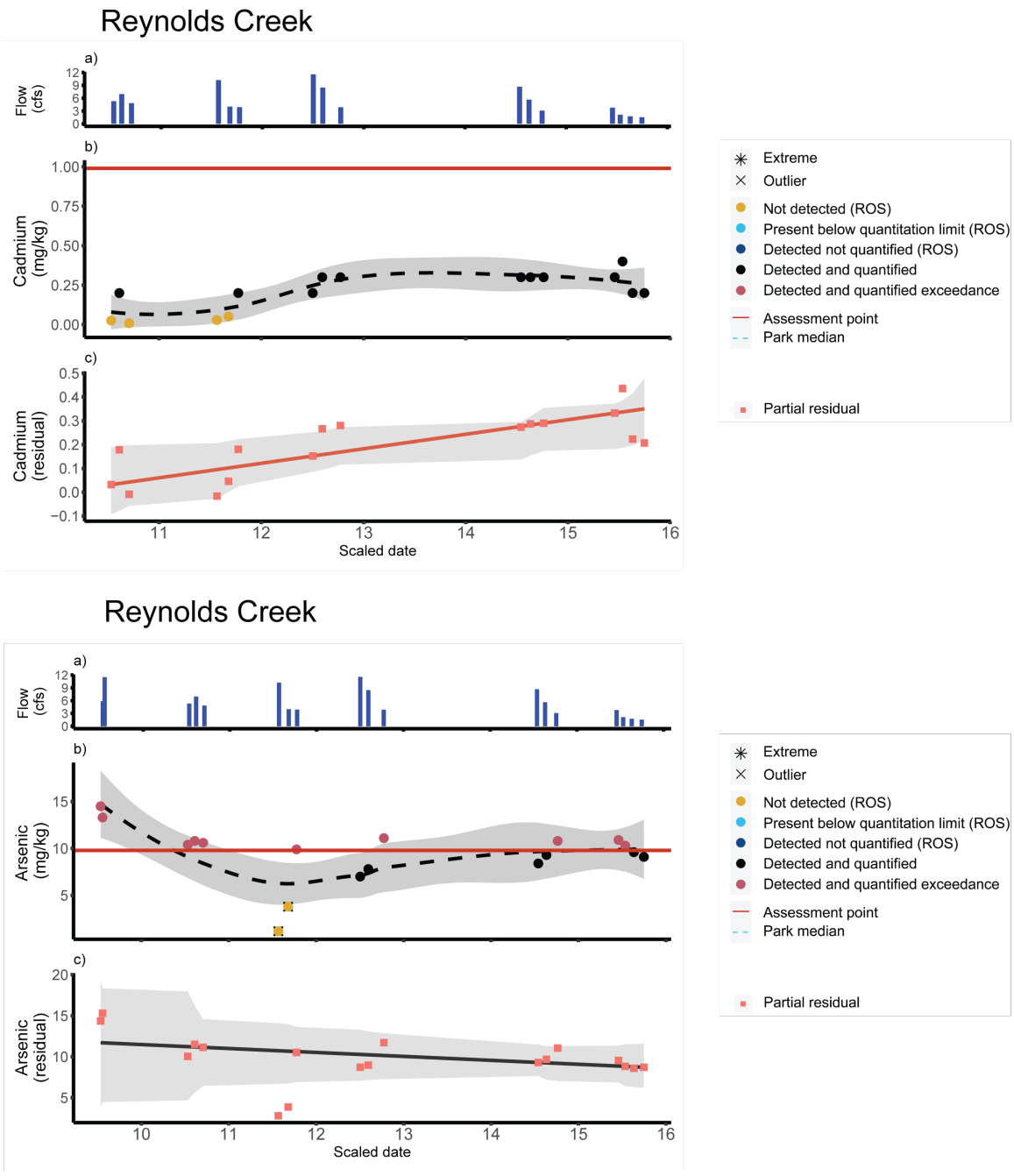


Figure 44. Select trend models for metals in sediment at the Reynolds Creek sentinel stream monitoring site in Glacier National Park from 2009 to 2019: (top) cadmium, (bottom) arsenic. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment point (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (significant for cadmium) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

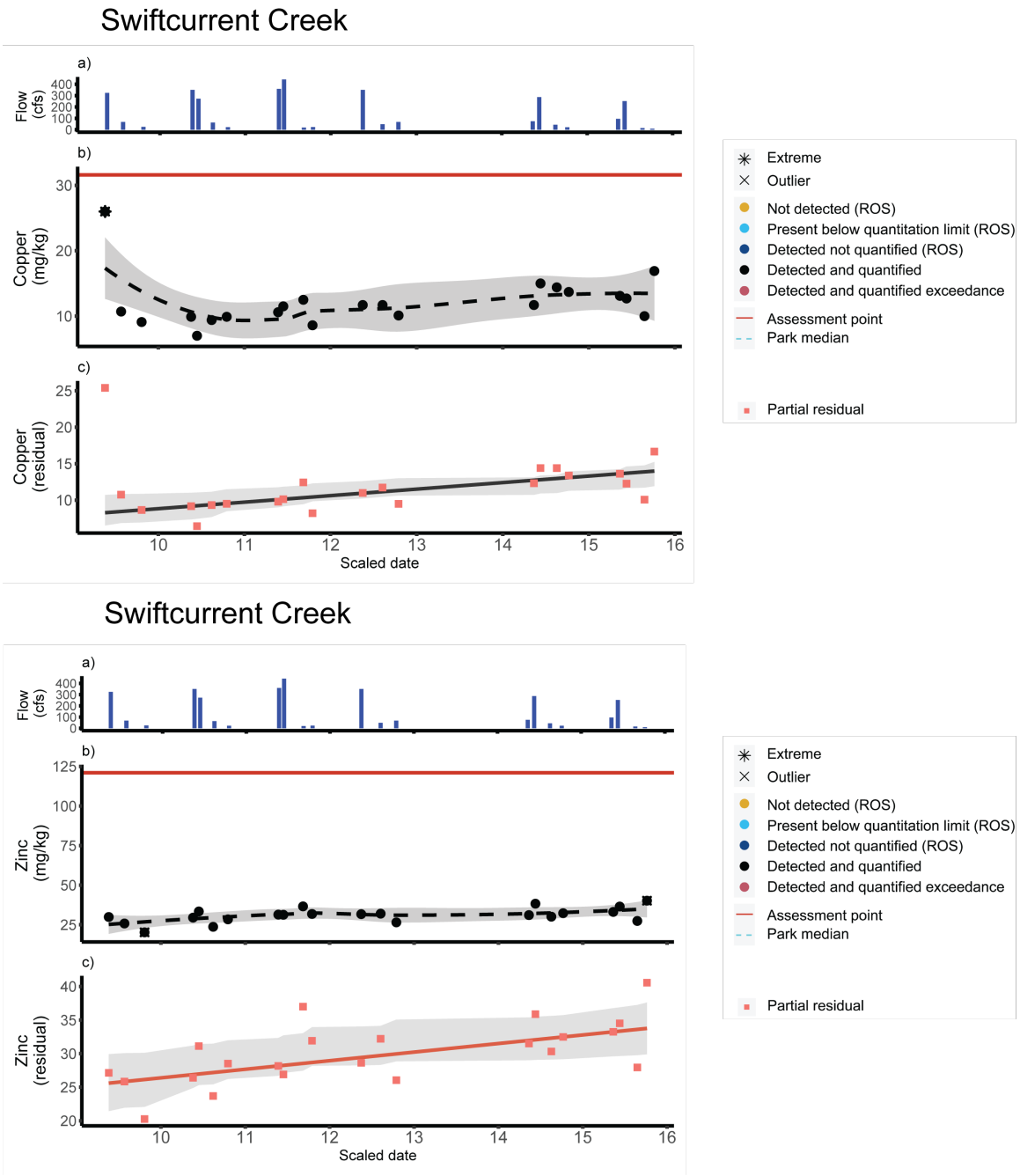


Figure 45. Select trend models for metals in sediment at the Swiftcurrent Creek sentinel site in Glacier National Park from 2009 to 2019: (top) copper and (bottom) zinc. (a) Stream flow at each sample event as directly measured or estimated from a water balance model of drainage in the site's watershed. (b) Raw concentrations at each sample event with a loess fit (black dashed line) and its 95% confidence intervals (gray shaded band). As available, assessment point (red solid line) and 2007–2009 park-scale median are shown. (c) Partial residuals and the fitted line (significant for zinc) of the response vs. scaled date from the main quantile regression model. Legend contains symbols that may not apply to this figure.

Parkwide 2007–2009 baseline and other previous results

We did not begin sampling metals in sediments until 2009 and thus do not have these results for the 2007–2009 baseline survey. Concentrations of metals in water in the 2007–2009 survey were very low, generally near or below their detection limits, suggesting no systemic metal in water problems in the park's streams. Importantly, a strong season effect usually occurs in the concentrations of metals in water, with peaks in April, May, and June, reflecting greater sediment transport by rivers during snowmelt (Mast 2007)—our base flow survey data may not have captured this well. Mast (2007) and Mills et al. (2011) also do not report any metals in sediment data. Both reports indicate that metals in water at specific sites were almost always below detection limits. Select and limited detections, always below any criteria, were likely attributed to natural sources, with the few detected concentrations highest during runoff, reflecting greater sediment transport by rivers during snowmelt. Neither report analyzed metals in water for trends.

QAQC results (in preparation by Schweiger and others) indicate that most lab and field metal in sediment duplicates had acceptable relative percent differences. The percentage of outliers and extremes on these data was also not excessive. There was some contamination in field blanks, but any paired sample with contamination was removed from all analyses. In general, our metal in sediment QAQC suggests these models can be interpreted without much concern for QAQC. Given the naturally low concentrations of metals in our data, there was a surprisingly low level of censoring in these results, with only cadmium and mercury exceeding the limits of our imputation methods to reliably impute values (we do not estimate trend models when this occurs).

Condition

Of the five metals analyzed, only arsenic had values above its assessment point. This happened at least once at three out of the six sites, with the percent of results greater than the assessment point, ranging from 5.4 to 55.6% of values. The high percentage (10 out of 18 results) was at Reynolds Creek. Our assessment point for arsenic (9.79 mg/kg) is a reference threshold, with arsenic concentrations greater than this typically interpreted as in an intermediate disturbance class. However, as we expect this assessment point is conservative for a stream like Reynolds Creek in the middle of a wilderness park, with no known local anthropogenic sources of metals, we interpret this as surprising and, if confirmed, potentially an emergent issue for the park. Of note, there was no significant trend in arsenic at Reynolds Creek, and the trend coefficient was negative, suggesting values may have been decreasing over our period of record.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead had median values of these metals in sediment that were comparable to other large river sites, and there were no exceedances in the limited data we have for these sites.

Discussion and management application

Increasing concentrations of many metals in the sediment of our sentinel sites are surprising. In addition, the level of exceedance at the Reynolds Creek site may be of potential concern to park management. There are no anthropogenic sources for these metals in the near pristine watershed of Reynolds Creek. There is seasonal visitor use in the drainage and some park facilities, but nothing that should be leading to these levels of arsenic nor an increasing trend in cadmium. Metals at these

sites are most likely from natural sources; however, they may be mobilized due to climate processes impacting acidity (see section 3.3.4 above) and other aspects of the chemical environment. Thies et al. (2017) document high metal concentrations in a watershed in the Alps with slightly acidic conditions (like Reynolds Creek) and no anthropogenic source of observed high metal concentrations. Todd et al. (2012) suggests that climate driven melting of permafrost accelerates weathering and sulfide oxidation of metals and is a probable explanation for increasing concentrations in a Colorado mountain watershed. They also suggest that the phenomenon is linked to water balance. Falling groundwater recharge rates from increased evapotranspiration can lead to lower water tables and more subsurface rock volume exposed to oxygen. This can potentially allow more total sulfide oxidation and both elevated sulfate and metal concentrations in groundwater discharging to streams. Our water balance models of the Reynolds Creek watershed (see section 3.3.4 above) do show a variable but marginal increase in summer evapotranspiration from 1980 to 2020.

Additional focused work should examine the potential sources and biological responses to these metals in Reynolds Creek and other sentinel sites in Glacier National Park as climate-induced degradation of water quality in watersheds draining mineralized areas has significant implications for the ecological integrity of these systems. Unfortunately, we did not anticipate this result and in 2015 we shifted to a 5-year cycle on sediment samples at all sites except on the North and Middle Forks of the Flathead River. Also, our period of record at Reynolds Creek is only 2009–2015. We have changed this and we began sampling sediment metals at all sites in Glacier in 2022.

3.4 Physical habitat

3.4.1 Introduction

The physical habitat of streams includes channel dimensions and pattern (e.g., width and depth, ratio of riffles to pools, gradient, etc.), sediment characteristics (the size and types of sediments make up the streambed and whether the channel is “armored”), amount and type of woody debris in the stream, and the amount and type of vegetation in the riparian zone. These characteristics are important determinants of the species of plants and animals that live in and near the stream, including diatoms, macroinvertebrates, and fish. Stream physical habitat can be negatively impacted by human stressors (Kaufmann et al. “Physical habitat in conterminous US streams and rivers, Part 1,” 2022; “Physical habitat in conterminous US streams and rivers, Part 2,” 2022)). Even in a wilderness park like Glacier National Park, where human use is relatively low (e.g., there are no dams), streams can be impacted by development such as roads and trails that produce sedimentation, or by logging outside the park that might change the delivery of large wood in a stream. Moreover, larger scale or indirect human disturbances such as climate change can change fire frequency and severity, which can increase variability in hydrology and change patterns of wood delivery to channels.

The SEI protocol adopts a suite of field methods (Figure 46) for habitat developed for regional surveys (Fitzpatrick et al. 1998; Kaufmann et al. 1999; Stoddard et al. 2005; Kaufmann et al. 2009; Heitke et al. 20008) that efficiently provide diverse, useful data. Most metrics generated from these data are precise and allow meaningful comparisons across large numbers of sites within a region (or a large park like Glacier). We base our selection of the 19 “core” metrics presented in this report on Stoddard et al. (2005) and Kaufmann (1993). The SEI database includes dozens of additional habitat

metrics (available upon request). The core attributes fall within six general categories of physical habitat, including volume, complexity, cover, sediment, floodplain interaction, and riparian vegetation. The attributes estimated by these metrics are considered important for evaluating physical habitat in multiple monitoring and assessment programs and are incorporated in many other habitat survey protocols (e.g., Platts et al. 1983; Lazorchak et al. 1998; Fitzpatrick et al. 1998; Angradi et al. 2006; Kusnierz et al. 2013).



Figure 46. Collecting residual pool depths on the North Fork Flathead—Glacier Rim sentinel stream monitoring site in Glacier National Park, 2019. NPS / B. SCHWEIGER

This report was largely completed before work presented by Kaufmann et al. ("Physical habitat in conterminous US streams and rivers, Part 1," 2022; "Physical habitat in conterminous US streams and rivers, Part 2," 2022) updated a list of suggested "best" metrics. Future work will review these and adjust our selection accordingly. We also do not use metrics scaled to watershed size, its land use composition (i.e., human disturbance), as in this more recent physical habitat work. This approach may be less relevant in a system like Glacier National Park, where we expect little human disturbance and thus less of a need to isolate it via control of geoclimatic context. We also do not make quantitative comparisons across sites that would require control of covariates.

3.4.2 Summary condition

Summary condition statements condense physical habitat results into concise overviews of key aspects of trends, assessment, and possible management applications or concerns (Table 16).

Table 16. Summary condition table for stream physical habitat at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Overall physical habitat	19 indicators	Physical habitat includes attributes like channel shape and size, sediment levels, and riparian vegetation and is the template upon which all ecological processes in the park's sentinel streams occur. Stream habitat can be impacted by small scale human disturbances, such as nearby park roads, and large scale processes like climate change, forest fires, and changing runoff patterns in watersheds inside and outside the park. Overall, trends from 2007 to 2019 in the condition of physical habitat at Glacier National Park sentinel sites showed few areas of concern. There was some indication of possible fine sediment issues and, using a conservative assessment point, the frequency of invasive/exotic plants was in a non-reference condition at many sites. Otherwise, the physical habitat indicators at the sentinel sites and over our period of record were generally stable and largely in reference condition. We have high confidence in this assessment.	
Volume	Residual pool depth	Habitat volume, or the amount of in-channel habitat available to stream biota, is an important determinant of the ecological condition of streams in the park. We estimate pools with a residual pool metric derived from the profile of the stream bottom along its deepest point that quantifies pools regardless of how much water is in the channel. There were no significant or marginally significant trends in residual pool depths at any sentinel site from 2007 to 2019, but more data on the larger rivers will help resolve patterns. There are no management, regulatory or known ecological assessment points for residual pool depths. We estimate an ecoregional assessment point from reference sites that provides general guidance for Glacier stream habitat volume under an expectation of a reference state. Very few habitat cover results were outside the reference condition suggested by the assessment point. Values approached the assessment point at Swiftcurrent Creek in 2018, perhaps due to lingering effects from a beaver dam in the sample reach and the ongoing draining of pools and channel bedform restructuring. The North Fork Flathead—Glacier Rim site had several values below the assessment point, but we suspect this is more due to methodological challenges than a human caused disturbance. We have high confidence in this assessment.	
Complexity	Thalweg depth variability	More complex stream habitat should generally support greater ecological integrity and provide for a more intact, higher functioning system. The coefficient of variation in the deepest part of the channel with the most flow (the thalweg) estimates how variable (or complex) the bottom of a stream is. There were no significant or marginally significant trends in coefficient of variation in thalweg depths at any sentinel site from 2007 to 2019. Increasing thalweg variability at the North Fork Flathead—Border site approached marginal significance. Note that this is an improving habitat condition should additional data confirm this trend. There are no management, regulatory, or known ecological assessment points for the coefficient of variation in thalweg depth. We estimate an ecoregional assessment point from reference sites that provides general guidance for stream habitat volume under an expectation of a reference state. Levels of variation in thalweg depths at sentinel sites from 2007 to 2019 were within the range seen at comparable ecoregional reference sites and we do not see any evidence of direct human driven impacts to habitat complexity at our sentinel sites that might be of concern to the park. We have high confidence in this assessment.	

Table 16 (continued). Summary condition table for stream physical habitat at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Complexity (continued)	Density of large wood	Woody debris is one of the most important sources of structure and complexity in many stream types. Large wood is critical for creating fish habitat, slowing down flows, and in its impact on sediments, stabilizing channels, moderating sediment transport, and increasing transient storage of sediment and nutrients. We use two metrics for wood. First, wood volume per length of stream/river channel can be related to wood inputs (from riparian) and transport out of the sample reach. It is strongly influenced by the size of the stream or river channel. Second, wood density is more directly experienced by biota as an element of habitat complexity and as a moderator of sediment transport and can more easily be related to sources of wood input (e.g., riparian tree cover). There were no significant or marginally significant trends in the density of large wood at any sentinel site from 2007 to 2019. We use an ecological threshold from USFWS developed for bull trout habitat assessments for our density of large wood assessment point. The threshold defines the density of wood needed for a properly functioning bull trout population. The bull trout habitat suitability threshold we apply to large wood density was not met at several sites, especially the new sentinel sites on the North Fork Flathead—Glacier Rim and the Middle Fork Flathead. While some potentially large wood densities that were lower than reference may warrant some caution, we do not see any evidence of direct human driven impacts to habitat complexity at our sentinel sites that might be of concern to the park. We therefore rate habitat complexity as measured by the density of large wood as in reference condition, but with medium confidence.	
	Wood volume	Woody volume is equally important to structure and complexity in most stream types. Smaller wood pieces, especially when they add up to more total volume, function much like large wood, particularly in smaller channels. There were no significant or marginally significant trends in total wood volume at any sentinel site from 2007 to 2019. There are no management, regulatory, or known ecological assessment points for wood volume. We estimate an ecoregional assessment point from reference sites that provides general guidance for Glacier National Park stream habitat volume under an expectation of a reference state. Wood volume was somewhat reduced at Swiftcurrent Creek, perhaps due to the hydrologic effect of the beaver dam or foraging activities by the beaver themselves as an entirely natural cause. However, at this time, we do not see any evidence of direct human driven impacts to habitat complexity as measured by wood volume at our sentinel sites that might be of concern to the park. We have high confidence in this assessment.	

Table 16 (continued). Summary condition table for stream physical habitat at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Cover	Areal cover of all natural features	We summarize habitat cover across all types of natural features such as large wood, brush, overhanging vegetation, live trees and roots, boulders, rock ledges, and undercut banks. Total natural cover did have two significant or marginally significant trends—increasing at Swiftcurrent Creek and decreasing at Reynolds Creek. We use a flat overall trend direction for the summary condition, given these offsetting patterns. There are no management, regulatory, or known ecological assessment points for total natural cover. We estimate an ecoregional assessment point from reference sites that provides general guidance for natural cover under an expectation of a reference state. Total natural cover was often lower than a reference state, especially at the North Fork Flathead sites. However, we do not see any evidence that these patterns are non-natural or suggestive of anthropogenic impacts of concern to the park. The beaver dam at Swiftcurrent Creek may have contributed to the increasing and higher natural cover. At Reynolds Creek, the causes behind fairly steep decline are uncertain. Exploratory models suggest interesting interactions among habitat cover, discharge, and watershed scale evapotranspiration. While some potentially lower-than-reference natural cover estimates may warrant some caution, additional data are needed. We retain a green or reference condition rating, but with medium confidence for the assessment.	
	Pool density	Pools provide critical cover for fish. We estimate pools using a threshold of 20 cm in residual depths as described for habitat volume (as measured by residual pool depth) above. There were no significant or marginally significant trends in pool density at any sentinel site from 2007 to 2019. We use an ecological threshold from USFWS developed for bull trout habitat assessments as our assessment point for pool density. We scale the assessment point to the measured stream width and sampled length at each site. The threshold defines a pool density needed for suitable bull trout habitat. The beaver dam at Swiftcurrent Creek from 2013 to ~2015 may have contributed to higher pool density when the dam was in place, or its effects may have lingered. There is usually a positive relationship between beaver influenced habitat and fish habitat quality. However, as an example of lower pool density at a site with reduced beaver and deep-water habitat, this is likely not cause for management concern; this dynamic is natural and beaver populations in the park are largely secure. Swiftcurrent Creek is often used by visitors from the adjacent campground who rearrange channel structure (by enthusiastic kids)—this might also impact some residual pool depths, but again not to a degree that the park should be concerned and curtail this form of visitor engagement. We have high confidence in this assessment.	

Table 16 (continued). Summary condition table for stream physical habitat at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Sediment	Embed-dedness	Stream substrate size, density, and distribution are major components of in-channel and near shore habitat and can be important determinants of overall stream condition. Embeddedness estimates the proportion of sediment particles larger than sand that are surrounded by sand, fines, or silt. A completely embedded piece of substrate has no interstitial space for aquatic organisms. There were no significant or marginally significant trends in embeddedness at any sentinel site from 2007 to 2019. We use an ecological threshold from USFWS developed for bull trout habitat assessments as our assessment point for embeddedness. The threshold defines an embeddedness above which suitable habitat is less likely. The two sites where embeddedness was measured (Ole and Swiftcurrent Creeks) had at least one value outside the assessment point. However, this assessment point is conservative and embeddedness is highly variable in time and space. We do not see these results as problematic. We have medium confidence in this assessment.	
	Fines	Fine sediment (< 0.06 mm) is a leading cause of water quality impairment in many watersheds with intensive human use. Negative impacts to streams from fines in the park may be localized, such as runoff from a road or smaller scale construction projects in the front country or from watershed scale changes in runoff or climate across all sentinel sites. We estimate the percentage of sediment size classes with an extensive and systematic "pebble count." All sites except Reynolds Creek had stable trends in fines. However, at Reynolds Creek, the trend was for or an improved condition (or less fines), and this approached a marginal significance level. We use an ecological threshold that defines habitat suitability thresholds for fines for eight macroinvertebrate taxa that occur in the park as an assessment point. A surprising number of sentinel site sample events have fines above this assessment point. This is a rigorous threshold and is likely a good approximation of a causal or structural relationship between macroinvertebrate habitat quality and smaller sediments in the park's streams. We therefore rate this indicator as a caution. Given the improving (vs. flat) trend at Reynolds Creek and the (historical) occurrence of high conservation value macroinvertebrates in this stream, we present Reynolds Creek separately from our other sites. However, we still retain the caution, given the number of results above the assessment point at the site. We also have lower confidence in the increasing trend result for Reynolds Creek given its nonsignificant probability value. At the other sites, however, a medium confidence is appropriate given the clearer flat trend lines.	Reynolds  
	Sand and fines	The potentially negative impact from smaller sediment described above includes particles up to sand in size (< 2 mm), with larger size classes more important in larger river systems or in high energy channel types. There were no significant or marginally significant trends in sand and fines at any sentinel site from 2007 to 2019. We use an ecological threshold that defines habitat suitability thresholds for sand and fines for eight macroinvertebrate taxa that occur in the park. A relative high percentage of sand and fines at our sentinel sites during our period of record were outside the ecological threshold for macroinvertebrate habitat suitability. This is a rigorous threshold and is likely a good approximation of a causal or structural relationship between macroinvertebrate habitat quality and smaller sediments in the park's streams. We have medium confidence in this assessment.	

Table 16 (continued). Summary condition table for stream physical habitat at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Sediment (continued)	Relative bed stability	Relative bed stability (RBS) measures the tendency of streambed particles to resist transport under a given stream flow. It compares the measured median substrate size in a streambed to the maximum theoretical substrate size carried during bankfull flow conditions. Species are adapted to natural streambed movement—too much or too little can cause stress and lead to reduced ecological integrity. Our indicator of RBS combines multiple measures and is a robust and integrative measure of sediment at our sentinel sites. There were no significant or marginally significant trends in RBS at any sentinel site from 2007 to 2019. There was a suggestion of some increased fining (more tendency for fine sediments) at Swiftcurrent Creek. These patterns need to be confirmed with more data, and this suggests a medium confidence in this result. Values of RBS can be either too low (finer, more unstable streambeds) or too high (coarser, more stable streambeds), and we define assessment points on both sides of the RBS gradient using ecoregional reference sites as there are no management, regulatory, or known ecological thresholds. While the percentages of fines and sands outside the assessment points may warrant some concern, most sites had relative bed stability in a reference range suggested by ecoregional assessment points. There was a suggestion of some increased fining (more tendency for fine sediments to move) at Swiftcurrent Creek, perhaps related to the beaver dam in place between 2014 and 2016 and how this affects sediment in channels. We have medium confidence in this assessment. Sediments are also an important piece of our bioassessment of condition and trend at our sentinel sites (see bioassessment, section 3.5). Overall, there was not a strong biological response to sediments in the macroinvertebrate assemblage. However, there was some indication from diatoms that sediments were decreasing at most sites, especially Swiftcurrent Creek. This suggests that higher levels of measured sediments as part of our habitat protocol did not translate to a meaningful impact to a site's biology, or that there may be a lag in the relationship.	
Floodplain interaction	Bankfull width / depth	Habitat features that affect the interaction of a stream with its floodplain are important in determining stream condition. We develop four indicators estimating this. The first, bankfull width to depth ratio, is an integrator of channel response to changes in sediment supply and transport, with important relationships to over-bank flooding, channel shading, and hyporheic (or underground) flow. There were no significant or marginally significant trends in bankfull width to depth ratio. The directions of slopes varied with site and indicator, but overall, this aspect of stream and floodplain interaction was stable over our period of record. There are no management, regulatory, or known ecological assessment points for bankfull width to depth ratio. We estimate an ecoregional assessment point from reference sites that provides general guidance for this indicator under an expectation of a reference state. No bankfull width to depth value at our sentinel sites was outside the ecoregional assessment point, suggesting that, for this indicator at our sites, stream channels had good interactions with their floodplains over our period of record. We have high confidence in this assessment.	

Table 16 (continued). Summary condition table for stream physical habitat at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

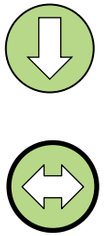
Vital sign	Indicator	Summary condition	Status, trend, and confidence
Floodplain interaction (continued)	Bankfull width / wetted width	Bankfull width to wetted width ratio also responds to changes in sediment supply and transport. It estimates the potential for inundation from floods. Channels with wider lateral bars and thus reduced ratios of bankfull width to wetted width lower the likelihood of smaller floods reaching into the floodplain. This is usually associated with lower stream condition. Bankfull width to wetted width at the North Fork Flathead—Border site marginally decreased. The net change was small, but in general, there was less distance from the river's wetted edge to its bankfull extent across the years of our sampling. This may indicate a decrease in complexity at the site. Other sites had no trend in this indicator. There are no management, regulatory, or known ecological assessment points for bankfull width to wetted width ratios. We estimate an ecoregional assessment point from reference sites that provides general guidance for this indicator under an expectation of a reference state. No bankfull width to wetted width value at our sentinel sites was outside the ecoregional assessment point, suggesting that, for this indicator at our sites, stream channels had good interactions with their floodplains over our period of record. We retain a good condition rating, even with the decreasing trend at the North Fork Flathead—Border site. We have high confidence in this assessment.	North Fork Flathead—Border 
	Incision	Stream incision occurs when the channel bed has dropped in elevation and the stream is disconnected from its floodplain. This leaves bankfull stage at a lower elevation than the top of the stream bank. It is usually caused by an increase of the erosive ability of a river, either from natural drivers or human stressors. There were no significant or marginally significant trends in incision at any sentinel site. The directions of slopes varied with site and indicator, but overall, this aspect of stream–floodplain interaction was stable over our period of record. There are no management, regulatory, or known ecological assessment points for incision. We estimate an ecoregional assessment point from reference sites that provides general guidance for this indicator under an expectation of a reference state. No incision value at our sentinel sites was outside the ecoregional assessment point, suggesting that, for this indicator at our sites, stream channels had good interactions with their floodplains over our period of record. Incision at Ole Creek did approach the assessment point in sample events that followed heavy spring runoff and a reworking of the channel, with large deposits of substrate into lateral bars and the creation of a complex bank structure, but this is a natural process. We have high confidence in this assessment.	

Table 16 (continued). Summary condition table for stream physical habitat at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Floodplain interaction (continued)	Sinuosity	Sinuosity is the ratio of channel length as the water flows divided by the straight-line distance between the two ends of a defined reach. The greater the sinuosity the slower water flows through a valley and the more chance it must interact with the terrestrial landscape. Sinuosity at Swiftcurrent Creek significantly increased over our period of record. The increase we see in our sinuosity metric corresponds with the beaver dam and may be because the channel downstream of the dam became dewatered, with some small-scale sinuous braiding, and this is reflected in our data. We do not interpret this as any sort of human disturbance effect (which usually straightens channels). Other sites had no trend in this indicator. There are no management, regulatory, or known ecological assessment points for sinuosity. We estimate an ecoregional assessment point from reference sites that provides general guidance for this indicator under an expectation of a reference state. No sinuosity value at our sentinel sites was outside the ecoregional assessment point, suggesting that, for this indicator at our sites, stream channels had good interactions with their floodplains over our period of record. We have high confidence in this assessment.	
Riparian vegetation cover and invasive / exotic plants	Riparian vegetation cover, all types	Riparian vegetation is important for channel structure, cover, shading, nutrient input, large woody debris, wildlife corridors, and as a buffer against anthropogenic disturbances. We use four indicators of riparian condition at the sentinel stream sties. First, total areal cover of riparian vegetation estimated visually from three height strata. There were no significant or marginally significant trends in total riparian vegetation cover at any site. This aspect of riparian vegetation was stable over our period of record. There are no management, regulatory, or known ecological assessment points for total cover. We estimate an ecoregional assessment point from reference sites that provides general guidance for this indicator under an expectation of a reference state. Reynolds Creek had total riparian vegetation cover below the reference range suggested by the assessment point. This is not surprising as alpine watersheds have no tree canopy and a reduced shrub layer. Concerns about the repeatability of some of the visual estimates we use are legitimate given the challenging conditions under which estimates are often made, the semi quantitative nature of visual cover estimation, and the high variability in vegetation cover. Signal:noise estimates from the 2007–2009 survey are likely a useful approximation of the quality of the data from the sentinel sites and all vegetation and canopy cover estimates had acceptable or even quite high signal:noise, suggesting they are rigorous data. We have high confidence in this assessment.	

Table 16 (continued). Summary condition table for stream physical habitat at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Riparian vegetation cover and invasive / exotic plants	Riparian vegetation cover, woody	We also visually estimate total areal cover of riparian woody vegetation as perhaps the most important structural element of the riparian zone. There were no significant or marginally significant trends in woody riparian vegetation cover at any site. This aspect of riparian vegetation was stable over our period of record. Of note, replicates across the years at the three wadeable sites had lower variability, but variance was higher across years at the North Fork Flathead—Border site, where it is harder to estimate canopy at some plots given the large river in the way. There are no management, regulatory, or known ecological assessment points for woody cover. We estimate an ecoregional assessment point from reference sites that provides general guidance for this indicator under an expectation of a reference state. No woody canopy cover results were below the assessment point, suggesting sentinel site riparian areas had sufficient woody cover over our period of record. We have high confidence in this assessment.	
	Total cover	We augment the visual estimates of cover with a quantitative method that estimates total cover of any feature just above the bank. This is important to capture other features besides vegetation that might shade a channel and to estimate low-to-the-ground cover, important in alpine watersheds. The canopy cover trend at Swiftcurrent Creek was significant, and trend at Reynolds Creek approached marginal significance. Both suggested that canopy cover increased over our period of record. The other two sites also had positive slopes in canopy cover but were stable. Exploratory models that incorporated estimates of production at sentinel sites saw an improvement in the estimate of trend in canopy cover. While we lack sufficient data to make firm conclusions, it may be that we see this increased production in higher canopy cover along the banks of our sentinel sites. Whether this is a positive, negative, or inconsequential pattern will require additional data. There are no management, regulatory, or known ecological assessment points for canopy cover. We estimate an ecoregional assessment point from reference sites that provides general guidance for this indicator under an expectation of a reference state. Cover was also reduced below the assessment point in several years on the North Fork Flathead—Border site. However, this is also not surprising given the wide lateral bars that push bankfull height away from the riparian corridor where there is heavy vegetation. We have medium confidence in this assessment.	

Table 16 (continued). Summary condition table for stream physical habitat at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Riparian vegetation cover and invasive / exotic plants	Invasive plants	Riparian systems are of special interest for monitoring status and trend in invasive/exotic plants for three reasons: they may be more likely to be invaded than upland ecosystems, the connectivity created by a stream network facilitates weed dispersal across a landscape, and many invasive/exotic species can alter stream ecosystems. Invasive/exotic species lists were developed with Glacier National Park staff in 2007 and have been monitored annually since. There were no significant or marginally significant trends in our Invasive Plant Index. We never observed an invasive/exotic plant at Reynolds Creek, so no trend model was possible there. Slopes in the Invasive Plant Index varied, but in general, the level of invasion at these sites was stable over our period of record. We do not estimate species-specific trend models, but Canada thistle, oxeye daisy, and spotted knapweed were the most common invasive nonnative species across all sentinel sites, with St. Johnswort and the meadow hawkweed complex abundant at Ole Creek. We use a Glacier National Park resource management defined/accepted assessment point of no occurrences of any of the target taxa at a sentinel stream site. This assessment point originated with the assumption that reference state is the absence of any exotic or nonnative species at a sentinel stream site in the park. This is in keeping with the NPS approach to managing invasive/exotic plants (NPS 2006b; see also Ringold et al. 2008). Park management recognized that this is likely not a realistic goal, but felt it best reflected the purpose of a wilderness park like Glacier. Nearly all years at all sites, except at Reynolds Creek, were above the park's conservative zero invasive/exotic plants assessment point. While the park's intention with setting this assessment point are reasonable, it is almost by definition going to lead to high percentages of values outside the reference condition. The overall level of invasive/exotic plants at these sites is not outside the norm for front country areas in the park. Ole Creek might have more than expected, but the site is on a trail that has a long history of stock use, prior to more strict controls of this activity. We have never recorded an invasive plant in the riparian corridor at Reynolds Creek. Alpine systems often have reduced levels of invasion given the short growing season and harsh climatic conditions. Pilot work on long term wetland monitoring by the Rocky Mountain Network in Glacier suggested changes in alpine hydrogeomorphic processes can trigger changes in vegetation composition by limiting establishment of disturbance-adapted species, such as *Salix* spp. (Salicaceae) that colonize bare mineral surfaces, thereby allowing the establishment of upland and exotic species. Possible climate change driven invasion of alpine systems certainly motivates ongoing monitoring. Given these concerns we have lower confidence in this assessment. Interestingly, the large sample of sites in the survey suggested a strong and intuitive relationship between human disturbance at a site and invasion by nonnative plants. Invasive plants also increased with the density of road–stream crossings in a site's watershed. With the smaller sentinel site sample size, spread across a decade, these relationships are no longer present. This suggests meaningful variability in the controls on invasion is spatial, not temporal (i.e., it is just a matter of time for invasives to get into the backcountry from the initial disturbed areas). There are some important caveats for our work with invasive/exotic plants. Our data on invasive/exotic riparian plants are relatively simple and do not replace efforts by Glacier to monitor invasive plants. SEI sites were not targeted towards areas with higher probability of invasive plants occurrence, and we likely miss local scale invasions. Finally, we do not conduct complete surveys of the vegetation in and around our sites because this would be cost and time prohibitive; therefore, we do not have data on the important community context of invasive plants.	

3.4.3 Volume

3.4.3.1 Background and relevancy

Habitat volume, or the amount of in-channel habitat available to stream biota, is an important determinant of the condition of streams (Kaufmann et al. 1999; Stoddard et al. 2005). The size of a stream is a function of several geoclimatic controls that are relatively immutable, such as its drainage area. However, anthropogenic activities can alter channel dimensions and flood or low flow discharge—all of which can alter the quantity and quality of aquatic habitat. While there are few direct human disturbances to stream habitat volume in Glacier National Park, over time, complex or indirect disturbances, such as increased sediment loading from more frequent fires or earlier spring runoff and more variable stream flows (such as are predicted with climate change, Goode et al. 2012), might alter stream habitat volume in the park.

Habitat data collected by the SEI protocol may be used to estimate relatively simple metrics of habitat volume, such as mean width or depth. While useful, these metrics are highly sensitive to daily variation in stream flow and may not necessarily reflect habitat volume during limiting drought conditions. Therefore, as a more informative indicator of habitat volume, we used a residual pool depth metric (Stack and Beschta 1989; Kaufmann et al. 1999). This metric is largely independent of stream discharge as it is based on the profile of the stream bottom, regardless of how much water is in the channel. Kaufmann et al. (1999) develops relatively simple methods (that we use) that generate reliable profile data without the need for actual surveyed elevations.

3.4.3.2 Assessment points

There are no management, regulatory, or known ecological assessment points for residual pool depths. We therefore estimate an ecoregional assessment point at the 5th percentile value from 75 ecoregional reference sites (Table 17). Values below or equal to this are in a non-reference condition. Residual pool depth decreases as condition in a stream deteriorates, as there is less habitat volume available for stream biota. Of note, when comparing residual depths among streams, it is important to scale metric values by basin size and channel slope, and then examine the relationship of these residuals to anthropogenic or climatic stressors (P. Kaufmann, US EPA hydrologist, pers. comm. 6/2023). However, fish don't necessarily experience "adjusted" or "scaled" habitat volume. Rather, they interact with the depths and areas of pools. Therefore, the habitat quality capacity of streams in Glacier National Park is related to the unadjusted values, while the response to human stress across sites is best examined on adjusted values (i.e., residuals).

Table 17. Select indicators of habitat volume at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. All parameters decrease as condition in a stream deteriorates. All AP types are ecoregional and are the 5th percentile value from 75 ecoregional reference sites. Values below or equal to the AP are in a non-reference condition.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Mean residual depth (cm)	North Fork Flathead—Glacier Rim	– [a]	– [a]	**66.7** [b]	1.1997	8/15/2008	8/20/2019	3	2.014	0.925 [c]	1.483	1.205	0.658	2.866
	North Fork Flathead—Border	−0.1164	0.694	0.0	1.1997	7/11/2008	8/22/2019	10	2.014	5.050	5.781	1.654	4.063	9.073
	Swiftcurrent Creek	−0.3585	0.438	0.0	1.1997	10/18/2007	8/22/2018	11	2.014	6.738	6.950	3.253	2.171	15.008
	Reynolds Creek	0.0459	0.889	0.0	1.1997	9/11/2008	8/21/2018	10	2.014	6.177	6.352	1.799	3.455	9.263
	Ole Creek	−0.2530	0.652	0.0	1.1997	8/27/2009	8/23/2018	9	2.014	7.342	7.826	2.921	4.24	13.705
	Middle Fork Flathead	– [a]	– [a]	0.0	1.1997	8/28/2018	8/19/2019	2	2.014	39.834 [c]	39.834	0.902	39.196	40.471

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] Outlier values. The value for the Middle Fork Flathead site is an outlier but it is correct as the site has two sections with very deep pools interspersed with more typical depths. Likewise for the North Fork Flathead—Rim site, but on the other extreme, with a very uniform bed profile.

3.4.3.3 Parkwide 2007–2009 baseline results

Schweiger et al. (2018) found a higher proportion than expected of the streams in the park had reduced residual pool volumes in a non-reference state (around 17%). However, this may have reflected many smaller streams with relatively "shallow" channel profiles included in the Glacier National Park survey relative to ecoregional reference sites and may not suggest that baseline residual pool volumes in the park were reduced due to meaningful human disturbances.

3.4.3.4 Trend

There were no significant or marginally significant trends in residual pool depth at any sentinel site (Figure 47). The directions of the slopes varied with site; however, three out of four sites did have a (nonsignificantly) decreasing amount of residual pool volume. This suggests stream volumes at our sentinel sites were stable from 2009 to 2019, with perhaps some likely unmeaningful decreases.

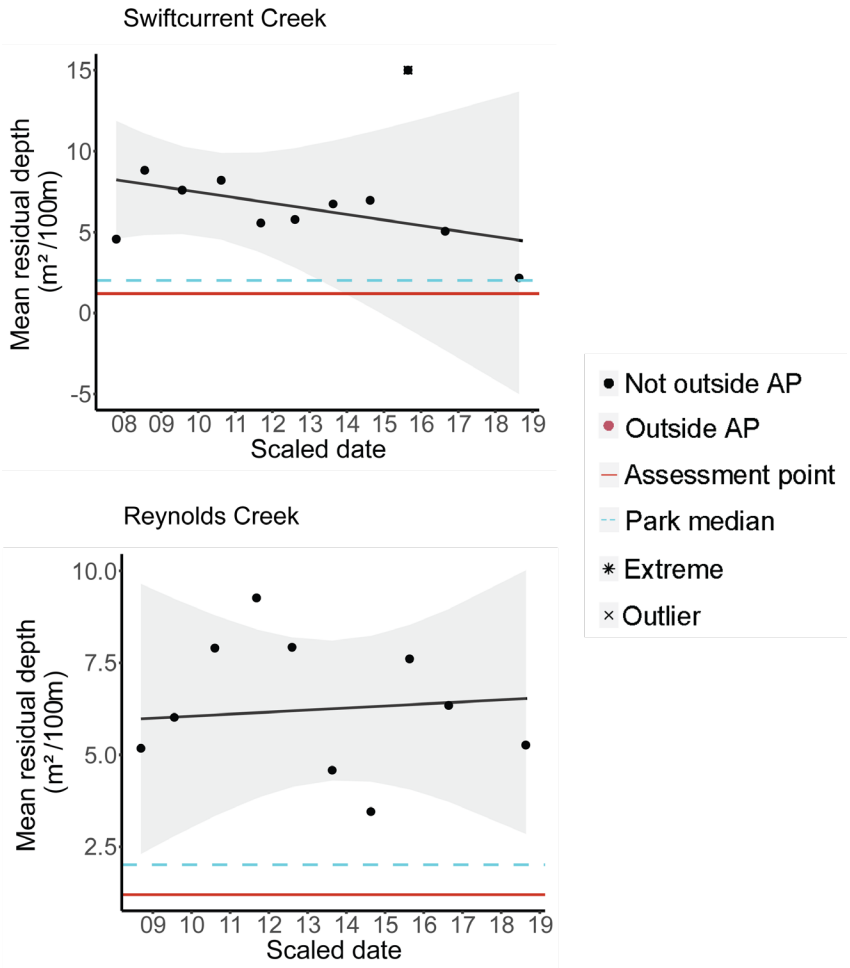


Figure 47. Select trend models for residual pool depth as an indicator of stream habitat volume in Glacier National Park from 2007 to 2019. (Top) Swiftcurrent Creek, (bottom) Reynolds Creek. As available, red horizontal lines show a non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (not a significant trend for either site) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

3.4.3.5 Condition

Very few residual pool volumes were outside a reference condition from the ecoregional assessment point, suggesting that, across time, Glacier National Park sentinel stream sites usually had residual pool volumes above the best sites in the surrounding ecoregions. They approached the assessment point at Swiftcurrent Creek in 2018, perhaps due to lingering effects from a beaver dam in the sample reach and the ongoing draining of pools and subsequent channel bedform restructuring when the dam was no longer in place. Two out of three results at the North Fork Flathead—Glacier Rim site were also below the assessment point, suggesting lower than reference pool volumes. This site is used for recreation and there is some low intensity land use in its watershed, but we suspect the potentially low pool volume at this site is more likely due to difficulty in measuring depths on a large river (Figure 46) or a possible mismatch in the ecoregional reference sites (where few sites are on larger rivers). Given the lower sample size and these complications, we do not interpret this as problematic. Large rivers can (naturally) have very high sediment transport rates, especially if they transport a lot of larger cobble substrate, and it is not surprising there were not a lot of deep pools in our big river sites (P. Kaufmann, US EPA hydrologist, pers. comm. 6/2023). Large wood can allow some big rivers to have deep pools, and we see these in our sample reaches, but these rivers' high power and mobile beds do not allow retention of small to medium size wood that might otherwise interact with flows to create pool habitat.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead had residual pool volumes somewhat unique from other results. The value for the Middle Fork Flathead site is an outlier, but it is correct as the site has two sections with very deep pools interspersed with more typical depths. Likewise, the value for the North Fork Flathead—Glacier Rim site is an outlier, but on the other extreme, with a very uniform bed profile.

3.4.3.6 Discussion and management applications

The ecoregional assessment point we use provides an estimate of stream volume in comparable stream sites that are likely in reference condition. It provides general guidance for Glacier stream habitat volume under an expectation of a reference state. Values outside this assessment point may be due to a variety of causes, and we do not see any evidence that any of these causes are non-natural or suggestive of anthropogenic impacts of concern to the park.

There are connections among sediment dynamics that contribute to reduced residual pool volumes (i.e., fine sediments infilling the tails of pools), land use, and climate change (Verdonschot et al. 2010), but without additional research, the results from our sentinel sites cannot make these links. Interestingly, at Swiftcurrent Creek, a large beaver dam that persisted in the site from 2014 or 2015 to sometime in 2016 may have initially spiked pool volumes but then reduced them when the dam broke and the channel likely was scoured, with sediment infilling parts of the channel profile. Processes like this are, of course, natural, and over time contribute to useful variation in stream habitat structure.

3.4.4 Complexity

3.4.4.1 Background and relevancy

More complex stream habitat supports a more intact, higher functioning stream system (Gorman and Karr 1978; Benson and Magnuson 1992). Stream monitoring in Glacier National Park provides several metrics that estimate this important attribute of streams (Kaufmann 1993; Stoddard et al. 2005). We selected three to interpret here: 1) coefficient of variation in thalweg depth, 2) the volume of all wood, and 3) the density of large wood (Table 18).

Table 18. Select indicators of habitat complexity indicators at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. All parameters decrease as condition in a stream deteriorates. AP types include ecological threshold (THR), and ecoregional (ERG). For ecoregional assessment point (ERG), we use the 5th percentile value from 75 ecoregional reference sites. The ecological threshold (THR) for large wood density is from USFWS (1998) for "Functioning Appropriately" bull trout populations. Values below or equal to the AP are in a non-reference condition.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Thalweg depth variability (CV, %)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	0.2344	ERG	8/15/2008	8/20/2019	3	0.400	0.347	0.326	0.037	0.283	0.348
	North Fork Flathead—Border	0.0090	0.169	0.0	0.2344	ERG	7/11/2008	8/22/2019	10	0.400	0.395	0.401	0.053	0.344	0.487
	Swiftcurrent Creek	0.0073	0.779	0.0	0.2344	ERG	10/18/2007	8/22/2018	11	0.400	0.307	0.394	0.235	0.251	1.056
	Reynolds Creek	0.0230	0.370	0.0	0.2344	ERG	9/11/2008	8/21/2018	10	0.400	0.481	0.574	0.302	0.346	1.387
	Ole Creek	0.0089	0.281	0.0	0.2344	ERG	8/27/2009	8/23/2018	9	0.400	0.308	0.319	0.046	0.262	0.383
	Middle Fork Flathead	– [a]	– [a]	0.0	0.2344	ERG	8/28/2018	8/19/2019	2	0.400	0.552	0.552	0.023	0.536	0.569
Density of large wood (#/100 m)	North Fork Flathead—Glacier Rim	– [a]	– [a]	**66.7** [b]	1.2	THR	8/15/2008	8/20/2019	3	18.600	0.455	1.818	2.362	0.455	4.545
	North Fork Flathead—Border	−0.2282	0.713	**30.0** [b]	1.2	THR	7/11/2008	8/22/2019	10	18.600	2.955	4.273	4.254	0	12.727
	Swiftcurrent Creek	−0.2473	0.241	18.2	1.2	THR	10/18/2007	8/22/2018	11	18.600	2.464	2.811	1.917	0.218	6.944

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value is ≥ 25%.

[c] Wood does not occur in alpine sites, so this assessment point is not relevant at Reynolds Creek.

[d] Significant at $p \leq 0.10$.

Table 18 (continued). Select indicators of habitat complexity indicators at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. All parameters decrease as condition in a stream deteriorates. AP types include ecological threshold (THR), and ecoregional (ERG). For ecoregional assessment point (ERG), we use the 5th percentile value from 75 ecoregional reference sites. The ecological threshold (THR) for large wood density is from USFWS (1998) for "Functioning Appropriately" bull trout populations. Values below or equal to the AP are in a non-reference condition.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Density of large wood (#/100 m) (continued)	Reynolds Creek	N/A [c]	N/A [c]	N/A [c]	N/A [c]	THR	9/11/2008	8/21/2018	10	18.600	0.000	0.048	0.152	0	0.481
	Ole Creek	0.8244	0.654	0.0	1.2	THR	8/27/2009	8/23/2018	9	18.600	19.311	17.933	9.929	1.242	34.442
	Middle Fork Flathead	– [a]	– [a]	**100.0** [b]	1.2	THR	8/28/2018	8/19/2019	2	18.600	0.455	0.455	0.643	0	0.909
Wood volume (log) (m^3/m^2)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	−3.9512	ERG	8/15/2008	8/20/2019	3	−2.000	−2.942	−2.774	0.650	−3.322	−2.056
	North Fork Flathead—Border	0.0225	0.830	0.0	−3.9512	ERG	7/11/2008	8/22/2019	10	−2.000	−1.973	−1.823	0.727	−2.516	0
	Swiftcurrent Creek	−0.00005	**0.0282** [d]	9.1	−3.9512	ERG	10/18/2007	8/22/2018	11	−2.000	−3.456	−3.613	0.509	−5.002	−3.144
	Reynolds Creek	N/A [c]	N/A [c]	N/A [c]	N/A [c]	ERG	9/11/2008	8/21/2018	10	−2.000	0.000	−0.428	1.354	−4.283	0
	Ole Creek	−0.0394	0.709	0.0	−3.9512	ERG	8/27/2009	8/23/2018	9	−2.000	−2.529	−2.449	0.622	−3.398	−1.214
	Middle Fork Flathead	– [a]	– [a]	0.0	−3.9512	ERG	8/28/2018	8/19/2019	2	−2.000	−1.373	−1.373	1.941	−2.745	0

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value is ≥ 25%.

[c] Wood does not occur in alpine sites, so this assessment point is not relevant at Reynolds Creek.

[d] Significant at $p \leq 0.10$.

The coefficient of variation in thalweg depth (where thalweg refers to the part of the channel with the most flow, usually the deepest location on a cross section) estimates how variable (or complex) the bottom of a stream is. Higher values (the range is 0 to ~1.0) suggest greater dispersion among thalweg depth readings (or a more complex channel profile). This metric will increase as flow stage declines or as streams are dewatered by hydrologic alteration or climate change. If we were comparing across sites, the metric would need to be scaled by watershed area (P. Kaufmann, pers. comm.). However, as we are focused on trend at individual sentinel stream sites, there was no need to scale it by size (Figure 48). Woody debris is one of the most important sources of structure and complexity in most stream types (Harmon et al. 1986; Robison and Beschta 1989; Gurnell et al. 2002). We used two woody debris metrics as estimates of the contribution of wood in the channel to habitat complexity: 1) the volume of woody debris per m^2 of bankfull channel, which incorporates both the frequency and amount of wood of any size, and 2) the density of large wood (> 0.3 m in diameter and > 5 m in length). Large wood is critical (i.e., Fausch and Northcote 1992), but smaller pieces, especially when they add up to more total volume, are also important, particularly in smaller channels (Lester et al. 2007). Given the distributional form of the wood volume metric data, we use a log transformation for this metric.



Figure 48. Field crew collecting data from a complex step-pool structured portion of the Reynolds Creek sentinel site in 2008, Glacier National Park. (NPS / BILLY SCHWEIGER)

3.4.4.2 Assessment points

There are no management, regulatory, or known ecological assessment points for variation in thalweg depth or wood volume, so we estimate an ecoregional assessment point at the 5th percentile value from 75 ecoregional reference sites (Table 18). These assessment points provide an estimate of complexity in comparable streams that are in likely reference condition; thus, they provide general guidance for park resource management's interpretation of stream habitat complexity at our sentinel sites under an expectation of a reference state.

We apply an ecological threshold developed for bull trout habitat assessments by USFWS (1998) to the density of large (> 0.3 m in diameter and > 5 m in length) wood scaled to 100 m of sampled stream. This threshold defines habitat conditions needed for a properly functioning bull trout fishery. The Reynolds Creek sentinel site naturally lacks large wood and is not a bull trout fishery, so we do not apply this assessment point to this site. As discussed above in the water temperature section, Swiftcurrent Creek is not an historical bull trout fishery given a large waterfall barrier just downstream from our site. However, we still use the bull trout assessment point for these two wood based complexity metrics, given that it is likely protective in general, and that Swiftcurrent Creek is representative of other streams on the park's east side that do have bull trout populations. Values below or equal to these assessment points are in a non-reference condition. All three of these indicators decrease as condition in a stream deteriorates, as there is less channel bedform variability or wood complexity available for stream biota.

3.4.4.3 Parkwide 2007–2009 baseline results

Schweiger et al. (2018) found high levels of thalweg variability, wood volume, and large wood density parkwide in 2007–2009. For the wood metrics, this was expected, given the lack of current logging and very little historical forestry in catchments that are internal to the park, as well as higher rates of delivery of woody debris into streams from productive watersheds and high gradient streams. We also expected natural and high levels of stream bottom complexity, given little to no anthropogenic disturbances to sediment or flow dynamics that largely control stream bedform.

3.4.4.4 Trend

Swiftcurrent Creek wood volume significantly increased, although with a very shallow slope. All other trends were non-significant in these three indicators (Table 18, Figure 49). The directions of the slopes varied with site and indicator. North Fork Flathead—Border site approached marginal significance. Note that should additional data confirm this trend, this is an improving habitat condition, and this may reflect reduced mineral exploration in the British Columbia watershed to the north of the park when the Province of British Columbia issued a Memorandum of Understanding banning energy resource development activity in the basin in 2010 (Province of British Columbia 2010; American Rivers 2010). Overall, these results suggest habitat complexity as estimated by these three measures at the four sentinel sites was stable from 2007 to 2019.

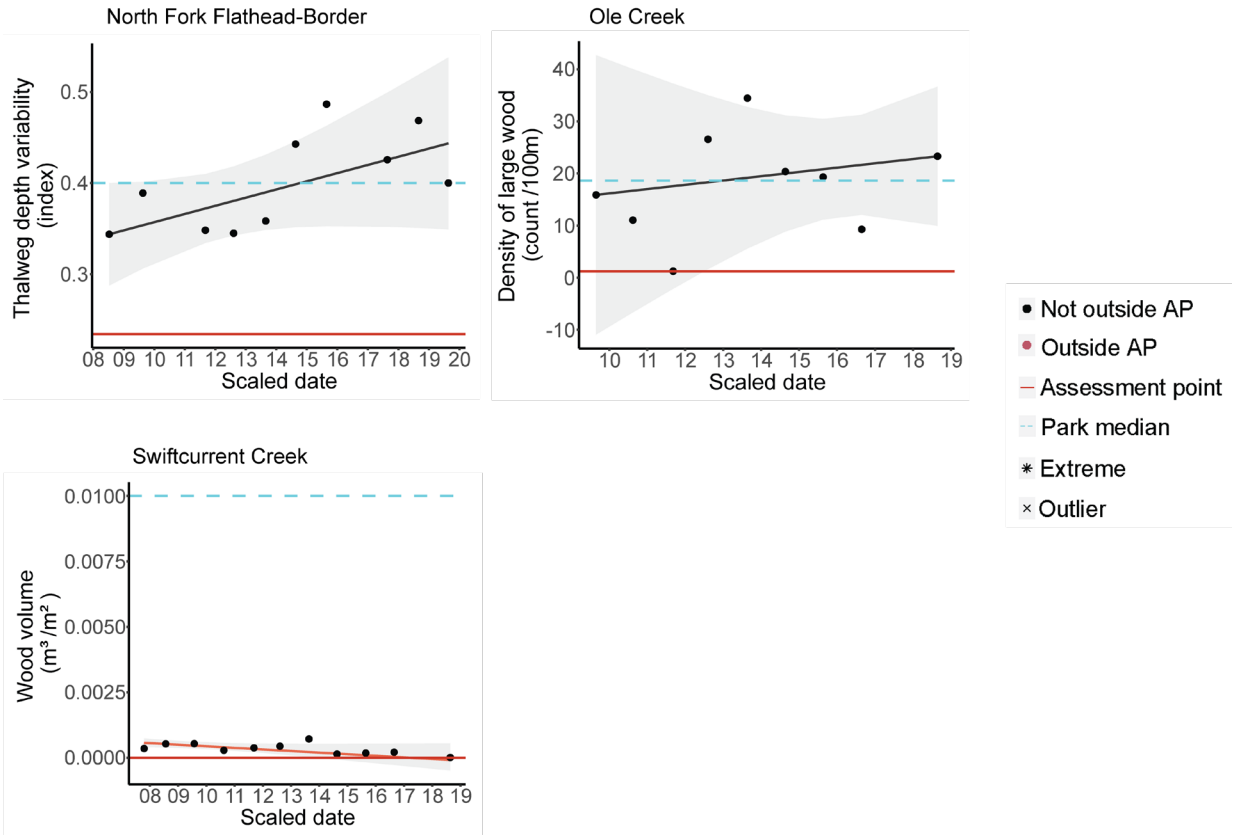


Figure 49. Select trend models for stream habitat complexity in Glacier National Park from 2007 to 2019. (Top left) thalweg depth variability at North Fork Flathead—Border, (top right) density of large wood at Ole Creek, and (lower left) wood volume (log) at Swiftcurrent Creek. As available, red horizontal lines show the non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (significant trend for wood volume) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

3.4.4.5 Condition

A few habitat complexity indicators were outside a reference range suggested by the assessment points we use. The bull trout habitat suitability threshold we apply to large wood density was often not met at several sites, especially the new sentinel sites on our large river sites on North Fork Flathead—Glacier Rim and the Middle Fork Flathead. It is unclear if this is real or a sampling effect—large wood is rare and the plots where it can be counted on a large river sample a small portion of the total area where wood might attach to the channel bed or banks. Additional large wood data in these rivers are needed. Saraiva et al. (2022) suggest that for the same wood input, wood density is lower on large rivers for two reasons: first, the wood volume is spread over a larger channel area; and second, larger streams more easily transport wood downstream. Large wood at Ole Creek (Figure 49) was almost always plentiful, suggesting for at least this aspect of stream habitat, the site is well suited to bull trout. Large wood density at Swiftcurrent Creek was sometimes lower than ideal. The beaver dam may have temporarily reduced recruitment of large woody debris into this site given lower velocity. (Although beavers do introduce wood of course via dams, this dam was made largely of smaller stems.) Total wood volume was also reduced at Swiftcurrent Creek, again

the hydrologic effect of the beaver dam or foraging activities by the beaver themselves may be the (entirely natural) cause here. Or this may all be an artifact of metric calculation, where the same amount of wood is averaged over the larger area created upstream of the beaver impoundment—more work is needed on this.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead had median thalweg depth variability like other sites' results. However, for both sites, the two wood metrics were meaningfully lower than other results. Wood on larger rivers may be naturally rarer than on other stream types, and until we have more data for these sites, and perhaps more relevant assessment points, we do not see any reason for concern.

3.4.4.6 Discussion and management applications

Several ecological processes, potentially reduced by human disturbance, contribute to stream habitat complexity, including wood recruitment, episodic high-flow stream channel restructuring events, land-use, and climate change, (i.e., Gordon and Meentemeyer 2006). However, we do not see any evidence of direct human driven impacts to habitat complexity at our sentinel sites that might be of concern to the park.

Beaver and their influence on habitat complexity at Swiftcurrent Creek are a useful example of natural drivers and how a careful and subtle interpretation of our results is necessary. Wood in streams, such as introduced by beaver, has been shown to be a useful mitigation tool for increasing variability in water temperature and lowering temperatures in select contexts (Sawyer and Cardenas 2012, but see Clark et al. 2021). In general, restoration of wood in channels can have many useful impacts on stream condition (Wohl et al. 2019), and, should issues with wood arise at sentinel sites or other streams in the park, resource management might include consideration of this in future planning.

3.4.5 Cover

3.4.5.1 Background and relevancy

Another key function of stream physical habitat is support of aquatic biota via physical cover as refugia (for example as protection from predators), as spawning habitat, or a variety of other protective functions. While many of the habitat metrics we estimate are related to this function, following Stoddard et al. (2005), Hankin and Reeves (1988), Kaufmann and Whittier (1997), and Kaufmann et al. ("Physical habitat in conterminous US streams and rivers, Part 2," 2022), we focus on two. These include: 1) the combined cover of all natural features (large wood, brush, overhanging vegetation, live trees and roots, boulders, rock ledges, and undercut banks) in the channel or on its banks, and 2) in-channel pool frequency (Table 19).

Table 19. Select indicators of habitat cover indicators at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. AP types include ecological threshold (THR), and ecoregional (ERG). For ecoregional assessment points (ERG), we use the 5th percentile value from 75 ecoregional reference sites. The ecological threshold (THR) for pool density is from USFWS (1998) for "Functioning Appropriately" bull trout populations, with values scaled to site specific counts by sampled reach length and width. All parameters decrease as condition in a stream deteriorates, and values below or equal to the AP are in a non-reference condition.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Areal cover of all natural features (proportion)	North Fork Flathead—Glacier Rim	– [a]	– [a]	**100.0** [b]	0.1182	ERG	8/15/2008	8/20/2019	3	0.450	0.082	0.088	0.010	0.082	0.1
	North Fork Flathead—Border	0.0036	0.839	**60.0** [b]	0.1182	ERG	7/11/2008	8/22/2019	10	0.450	0.086	0.135	0.140	0.005	0.466
	Swiftcurrent Creek	0.0125	**0.032** [c]	0.0	0.1182	ERG	10/18/2007	8/22/2018	11	0.450	0.309	0.301	0.066	0.214	0.418
	Reynolds Creek	−0.0405	**0.065** [c]	20.0	0.1182	ERG	9/11/2008	8/21/2018	10	0.450	0.245	0.234	0.118	0.041	0.443
	Ole Creek	−0.0103	0.603	0.0	0.1182	ERG	8/27/2009	8/23/2018	9	0.450	0.345	0.329	0.092	0.195	0.439
	Middle Fork Flathead	– [a]	– [a]	0.0	0.1182	ERG	8/28/2018	8/19/2019	2	0.450	0.133	0.133	0.008	0.128	0.139
Pool density (count per 100 m of stream length)	North Fork Flathead—Glacier Rim	– [a]	– [a]	**100.0** [b]	8.82	THR	8/15/2008	8/20/2019	3	N/A [d]	2.000	1.667	1.528	0	3
	North Fork Flathead—Border	−0.3356	0.276	**40.0** [b]	5.99	THR	7/11/2008	8/22/2019	10	N/A [d]	6.500	6.200	1.687	4	9

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] Trend is significant (p ≤ 0.10).

[d] N/A = We lack a park scaled pool density median value.

[e] N/A = Reynolds Creek is not a bull trout fishery, and we omit assessment using a pool density-based AP for this site.

Table 19 (continued). Select indicators of habitat cover indicators at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. AP types include ecological threshold (THR), and ecoregional (ERG). For ecoregional assessment points (ERG), we use the 5th percentile value from 75 ecoregional reference sites. The ecological threshold (THR) for pool density is from USFWS (1998) for "Functioning Appropriately" bull trout populations, with values scaled to site specific counts by sampled reach length and width. All parameters decrease as condition in a stream deteriorates, and values below or equal to the AP are in a non-reference condition.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Pool density (count per 100 m of stream length) (continued)	Swiftcurrent Creek	−0.2015	0.594	18.2	2.67	THR	10/18/2007	8/22/2018	11	N/A [d]	4.000	4.000	1.732	1	6
	Reynolds Creek	0.0000	1.000	N/A [e]	N/A [e]	THR	9/11/2008	8/21/2018	10	N/A [d]	6.000	5.200	2.200	0.002	8
	Ole Creek	0.0000	1.000	11.1	2.39	THR	8/27/2009	8/23/2018	9	N/A [d]	5.000	4.556	1.333	2	6
	Middle Fork Flathead	– [a]	– [a]	0.0	5.96	THR	8/28/2018	8/19/2019	2	N/A [d]	7.000	7.000	0.000	7	7

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] Trend is significant ($p \leq 0.10$).

[d] N/A = We lack a park scaled pool density median value.

[e] N/A = Reynolds Creek is not a bull trout fishery, and we omit assessment using a pool density-based AP for this site.

Natural feature cover is a mean visual estimate across eight type categories. We estimate pools using residual thalweg depths rather than a more subjective visual macrohabitat classification approach, which can miss small scale patterns in slack or pooled water. A residual depth profile approach also has the distinct advantage of being independent of flow stage. Consequently, pool metrics derived from residual depths are more repeatable than visual judgements of pools. Residual depth is the depth that, if flow were reduced to zero, water would still fill a depression in the channel profile. To define what a "pool" is in terms of residual depth, we use data from 46 sentinel site sample events in Glacier National Park from 2007 to 2019. The average stream depth in these was 64.03 cm. We assume that a meaningful pool that would provide cover for a typical salmonid is at least a third of this average depth. We therefore use a 20 cm residual depth (about 33% of 64 cm) as the estimate of what constitutes a pool in the bottom profile of a sentinel site channel. The final metric is expressed as counts per 100 m of stream length. We do not set a longitudinal distance requirement on a residual depth-based pool, but at a minimum, given how we space thalweg depth measurements, this will be at least 1.5 m—likely more than enough distance for a typical fish to find peace within.

3.4.5.2 Assessment points

Natural cover (Hughes and Vadas Jr. 2021) and pool frequency (Lisle and Hilton 1992) both decrease as condition in a stream deteriorates and values below the assessment points are in a non-reference condition. There are no management, regulatory, or known ecological assessment points for natural stream cover that we feel are relevant for Glacier National Park (but see Kaufmann et al., "Physical habitat in conterminous US streams and rivers, Part 1," 2022; Kaufmann et al., "Physical habitat in conterminous US streams and rivers, Part 2," 2022), so we estimate an ecoregional assessment point at the 5th percentile value from 75 ecoregional reference sites (Table 19). Ecoregional assessment points provide an estimate of cover in comparable stream sites that are in likely reference condition, and they provide general guidance for Glacier resource management considering stream habitat cover at these sites under an expectation of a reference state.

For pool density, given well-documented links with salmonid habitat suitability (i.e., Muhlfeld and Bennett 2001; Harper and Farag 2004) and land management practices (Lisle and Hilton 1991; Kershner et al. 2004), we use an ecological threshold from USFWS (1998) that is applied to biological assessments by agencies like NPS. We must assume that whatever was a pool in USFWS (1998)—and it is not entirely clear in the document—is functionally equivalent to our characterization of pools. We then calculate site specific assessment points (Table 19) that apply the mean measured stream width and sampled length at each site to the USFWS pool threshold, rescaling this to a total expected density of pools in each sentinel site (Table 19). Note that as with other bull trout-based assessment points, we do not assess Reynolds Creek site, which is not a bull trout habitat, but we do apply it to Swiftcurrent Creek, even though the site is not considered bull trout habitat by the park due to a fish barrier.

3.4.5.3 Parkwide 2007–2009 baseline results

Schweiger et al. (2018) found high levels of natural cover types in 2007–2009, with little suggestion of any meaningful parkwide anthropogenic effects. Only two hand-picked gradient sites had any indication of degraded habitat cover conditions due to human use (gradient sites were selected

largely to assess human disturbance). Similarly, about 70% of the streams in the park had pool density above a bull trout threshold. Lower pool density often occurred at sites unsuitable for bull trout populations for a variety of other reasons (elevation, stream size).

3.4.5.4 Trend

Total natural cover did have two significant trends—increasing at Swiftcurrent Creek and decreasing at Reynolds Creek (Table 19, Figure 50). There were no significant or marginally significant trends in pool density at any sentinel site (Table 19, Figure 51). However, for the North Fork Flathead—Border and Swiftcurrent Creek sites, the direction of these slopes was negative.

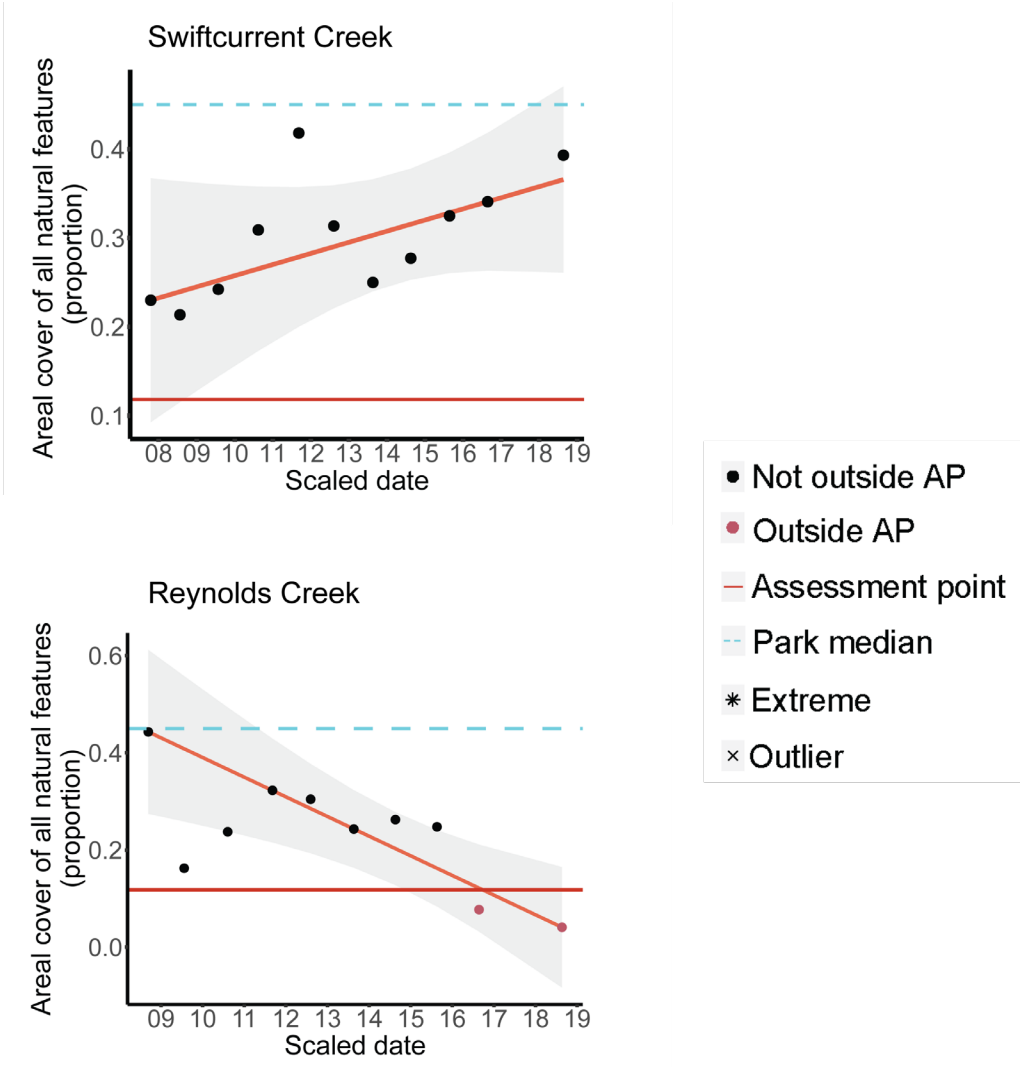


Figure 50. Cover of all natural feature types as an indicator of stream habitat cover in Glacier National Park from 2007 to 2019. (Top) Swiftcurrent Creek and (bottom) Reynolds Creek. As available, red horizontal lines show the non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (significant trend at both sites) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

3.4.5.5 Condition

Few natural cover results were outside the reference condition derived from the ecoregional assessment point, suggesting that, across time, most sentinel stream sites in Glacier National Park usually had cover above the best sites in the surrounding ecoregions. Natural cover was lower than the reference state on the North Fork Flathead—Border site 60% of the time. Lower cover may be due to a lack of overhanging vegetation in the littoral zone on this large floodplain river, where the banks at bankfull width are often sandy lateral bars and thus not conducive to much vegetation growth that overhangs the wetted channel. This could suggest that a big river scaled assessment point is needed for systems like the North Fork of the Flathead River. Pool density was lower than the reference state on the North Fork Flathead—Border site 40% of the time (Figure 51). This may need more investigation, but fewer deep pools (and lack of pool riffle structure in general) is expected in bigger rivers transporting a large amount of cobble and gravel bedload, such as the North Fork of the Flathead.

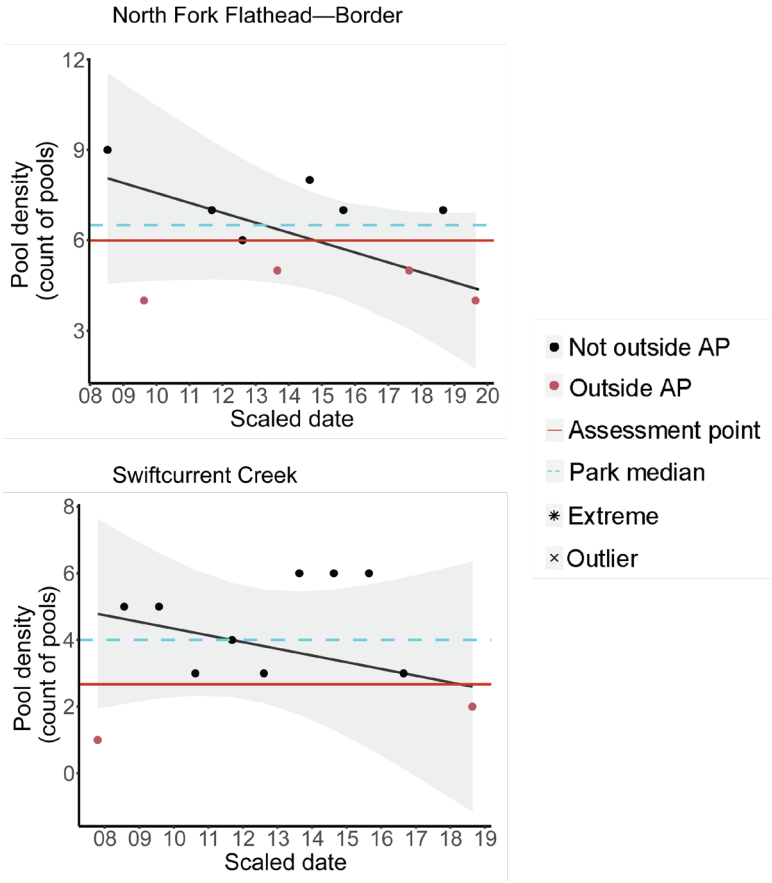


Figure 51. Pool frequency as an indicator of stream habitat cover in Glacier National Park from 2007 to 2019. (Top) North Fork Flathead—Border, and (bottom) Swiftcurrent Creek. As available, red horizontal lines show the non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (neither creek had significant trends) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead had natural cover or pool density results like other site results. However, a higher percentage of these results for both sites were below the assessment point, suggesting lower cover. Until we have more data for these sites, however, and perhaps more relevant assessment points, we do not see any reason for concern.

3.4.5.5 Discussion and management applications

The beaver dam at Swiftcurrent Creek from 2013 to ~2015 may have contributed to higher natural cover, with increased wood, overhanging vegetation, and deeper water. At Reynolds Creek the causes behind steep decline are uncertain. There are higher scores for individual cover elements (boulder and algae) earlier in the period of record. Crew scoring of cover may depend on stream flow, with higher flow likely reducing the quality of some observations. Flow did slightly increase at Reynolds Creek over the years and exploratory models suggest interesting interactions among habitat cover, discharge, and watershed scale evapotranspiration. Additional work is needed to resolve this, but we do not feel either trend represents concerns for park management.

Pool density at Swiftcurrent Creek was also highest during the years when a beaver dam was present or impacting the channel, but then fell to pre-dam levels. There is a large body of literature on the usually positive relationships between beaver influenced habitat and fish (i.e., Smith and Mather 2013). Lower pool density in years where beavers were not creating deep water habitat is likely not cause for management concern. The site is also used by visitors from the adjacent campground, with often notable rearrangement of the channel by enthusiastic kids that sometimes create pools in the thalweg (we do not recommend the park curtail this activity, but it is interesting that it might have an effect that appears in our data).

We do not see any evidence that any of these causes are non-natural or suggestive of anthropogenic impacts of concern to the park. As with other types of habitat indicators, the role of beaver at Swiftcurrent Creek is a useful example of how natural processes create a need for careful and subtle interpretation of these types of results. There are connections among the ecological processes that contribute to reduced stream habitat cover (i.e., channel bedform rearrangement), land use, and climate change (i.e., Townsend et al. 2004); however, data from our sentinel sites cannot conclusively make these links. Climate driven changes to stream discharge, runoff, and evapotranspiration likely will become more important (Verdonschot et al. 2010) in their influence on small scale responses, like fish cover, and monitoring of these responses will be important into the future.

3.4.6 Sediment

3.4.6.1 Background and relevancy

Stream substrate size, density, and distribution are major components of in-channel and near shore habitat and can be important determinants of overall stream condition (Figure 52). Along with bedform, substrate size influences hydraulic roughness, water velocity, and the interstices amongst substrate particles that provide habitat for macroinvertebrates and smaller vertebrates. The size of substrates and their relative extents on streambeds are often sensitive indicators of the effects of

human activities on streams (MacDonald et al. 1991; Barbour et al. 1999). Accumulations of fine substrate particles can fill interstitial spaces and reduce habitat for benthic fish, macroinvertebrates, and diatoms (Platts et al. 1983; Hawkins et al. 1983; Rinne 1988; Yamada and Nakamura 2002; Suttle et al. 2004). In addition, fines impede circulation of oxygenated water into hyporheic habitats. Decreases in particle size can destabilize stream channels (Wilcock 1997, 1998) and may indicate increases in the rates of upland erosion and sediment supply from anthropogenic disturbances (Lisle 1982; Dietrich et al. 1989; Waters 1995).



Figure 52. Measuring sediment at Ole Creek in 2014, Glacier National Park. (NPS / BILLY SCHWEIGER)

We include three relatively simple substrate focused metrics following Stoddard et al. (2005), and Kaufmann et al. (1999): 1) percent silt and finer ("fines"; < 0.06 mm), 2) percent sand and finer (< 2.0 mm), and 3) percent embeddedness (Table 20). Embeddedness estimates the proportion of sediment particles larger than sand that are surrounded by sand, fines, or silt. A completely embedded piece of substrate has no interstitial space for aquatic organisms.

Table 20. Select indicators of sediment at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. If more than one AP is used, the second AP is in parentheses. AP types include ecological threshold (THR), and ecoregional (ERG). All four metrics increase as condition in a stream deteriorates. Relative bed stability (RBS) also has both a lower and an upper AP; we use the 5th percentile and 95th value from 75 ecoregional reference sites, respectively. For relative bed stability, values below or equal to the lower AP are in a non-reference condition and values above or equal to the second AP are in a non-reference condition. The ecological threshold (THR) AP for fines and sand indicators are from Bryce et al. (2010) and were developed based on eight sediment sensitive macroinvertebrate taxa that occur in or around Glacier National Park. The ecological threshold (THR) for embeddedness is from USFWS (1998) for "Functioning Appropriately" bull trout populations. * Embeddedness is not measured at large river sites.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Embeddedness * (%)	Swiftcurrent Creek	1.3739	0.380	**63.6** [b]	20	THR	10/18/2007	8/22/2018	11	19.000	24.167	24.811	10.164	7.982	45.782
	Reynolds Creek	−0.8135	0.502	N/A [d]	N/A [d]	THR	9/11/2008	8/21/2018	10	19.000	14.681	14.640	6.305	3.58	24.764
	Ole Creek	1.2336	0.363	**11.1** [b]	20	THR	8/27/2009	8/23/2018	9	19.000	11.745	14.196	6.612	5.982	27.782
Fines (%)	North Fork Flathead—Glacier Rim	– [a]	– [a]	**33.3** [b]	3	THR	8/15/2008	8/20/2019	3	1.900	0.000	1.010	1.750	0	3.03
	North Fork Flathead—Border	N/A [c]	N/A [c]	**30.0** [b]	3	THR	7/11/2008	8/22/2019	10	1.900	0.000	1.515	2.575	0	6.061
	Swiftcurrent Creek	0.0913	0.857	**45.5** [b]	3	THR	10/18/2007	8/22/2018	11	1.900	2.222	3.165	3.467	0	11.111
	Reynolds Creek	−0.3141	0.174	**60.0** [b]	3	THR	9/11/2008	8/21/2018	10	1.900	3.396	3.191	1.608	0	5
	Ole Creek	−0.1533	0.686	22.2	3	THR	8/27/2009	8/23/2018	9	1.900	2.098	1.926	1.780	0	5.147

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] N/A There is insufficient variation in fines at North Fork Flathead—Border for quantile regression model estimation.

[d] N/A The USFWS (1998) AP for embeddedness does not apply to Reynolds Creek as this site is not a viable stream for bull trout.

Table 20 (continued). Select indicators of sediment at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. If more than one AP is used, the second AP is in parentheses. AP types include ecological threshold (THR), and ecoregional (ERG). All four metrics increase as condition in a stream deteriorates. Relative bed stability (RBS) also has both a lower and an upper AP; we use the 5th percentile and 95th value from 75 ecoregional reference sites, respectively. For relative bed stability, values below or equal to the lower AP are in a non-reference condition and values above or equal to the second AP are in a non-reference condition. The ecological threshold (THR) AP for fines and sand indicators are from Bryce et al. (2010) and were developed based on eight sediment sensitive macroinvertebrate taxa that occur in or around Glacier National Park. The ecological threshold (THR) for embeddedness is from USFWS (1998) for "Functioning Appropriately" bull trout populations. * Embeddedness is not measured at large river sites.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Fines (%) (continued)	Middle Fork Flathead	– [a]	– [a]	0.0	3	THR	8/28/2018	8/19/2019	2	1.900	0.000	0.000	0.000	0	0
Sand and fines (%)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	10	THR	8/15/2008	8/20/2019	3	3.900	0.000	0.505	0.875	0	1.515
	North Fork Flathead—Border	−0.1363	0.640	0.0	10	THR	7/11/2008	8/22/2019	10	3.900	2.273	2.441	2.024	0	4.545
	Swiftcurrent Creek	−0.0570	0.941	**63.6** [b]	10	THR	10/18/2007	8/22/2018	11	3.900	11.111	11.816	5.364	3.306	21.429
	Reynolds Creek	−0.2544	0.694	10.0	10	THR	9/11/2008	8/21/2018	10	3.900	6.186	6.741	3.717	0.752	14.407
	Ole Creek	−0.9841	0.258	22.2	10	THR	8/27/2009	8/23/2018	9	3.900	8.633	7.594	4.153	1.575	13.971
	Middle Fork Flathead	– [a]	– [a]	0.0	10	THR	8/28/2018	8/19/2019	2	3.900	1.591	1.591	0.107	1.515	1.667

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] N/A There is insufficient variation in fines at North Fork Flathead—Border for quantile regression model estimation.

[d] N/A The USFWS (1998) AP for embeddedness does not apply to Reynolds Creek as this site is not a viable stream for bull trout.

Table 20 (continued). Select indicators of sediment at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. If more than one AP is used, the second AP is in parentheses. AP types include ecological threshold (THR), and ecoregional (ERG). All four metrics increase as condition in a stream deteriorates. Relative bed stability (RBS) also has both a lower and an upper AP; we use the 5th percentile and 95th value from 75 ecoregional reference sites, respectively. For relative bed stability, values below or equal to the lower AP are in a non-reference condition and values above or equal to the second AP are in a non-reference condition. The ecological threshold (THR) AP for fines and sand indicators are from Bryce et al. (2010) and were developed based on eight sediment sensitive macroinvertebrate taxa that occur in or around Glacier National Park. The ecological threshold (THR) for embeddedness is from USFWS (1998) for "Functioning Appropriately" bull trout populations. * Embeddedness is not measured at large river sites.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Relative bed stability (log, index)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	0.144 (-2.047)	ERG	8/15/2008	8/20/2019	3	−0.762	−0.483	−0.536	0.099	−0.65	−0.474
	North Fork Flathead—Border	0.0199	0.458	0.0	0.144 (-2.047)	ERG	7/11/2008	8/22/2019	10	−0.762	−0.403	−0.385	0.196	−0.639	0.013
	Swiftcurrent Creek	−0.0475	0.320	0.0	0.144 (-2.047)	ERG	10/18/2007	8/22/2018	11	−0.762	−0.803	−0.796	0.421	−1.754	−0.192
	Reynolds Creek	0.0399	0.455	0.0	0.144 (-2.047)	ERG	9/11/2008	8/21/2018	10	−0.762	−0.482	−0.532	0.313	−1.016	−0.147
	Ole Creek	−0.0289	0.606	0.0	0.144 (-2.047)	ERG	8/27/2009	8/23/2018	9	−0.762	−0.935	−0.881	0.276	−1.215	−0.351
	Middle Fork Flathead	– [a]	– [a]	**100.0** [b]	0.144 (-2.047)	ERG	8/28/2018	8/19/2019	2	−0.762	0.476	0.476	0.026	0.458	0.494

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] N/A There is insufficient variation in fines at North Fork Flathead—Border for quantile regression model estimation.

[d] N/A The USFWS (1998) AP for embeddedness does not apply to Reynolds Creek as this site is not a viable stream for bull trout.

Relative bed stability

We also use relative bed stability (RBS) as an integrative and important metric (Kaufmann et al. 2008; Kaufmann et al., "Physical habitat in conterminous US streams and rivers, Part 1," 2022; Kaufmann et al., "Physical habitat in conterminous US streams and rivers, Part 2," 2022) to characterize substrates in Glacier National Park. RBS measures the tendency of streambed particles to resist transport under a given stream flow. RBS compares the measured median substrate size in a streambed to the maximum theoretical substrate size carried during bankfull flow conditions. This ratio then estimates the deviation from what a site "should" have under minimal stress or reference conditions. The ratio can indicate that a site either has too many fine sediments or too many coarse sediments. For example, if a stream can move large boulders during bankfull flow events, yet the measured median substrate is fines and silts, an RBS metric would suggest that excessive fine sediment loading is present (or the site is "fining"). In contrast a site with too many coarse sediments is "armoring." RBS is a key component of habitat structure for aquatic organisms. Species are adapted to natural streambed movement—too much or too little can cause stress and led to reduced ecological integrity. Because RBS depends on both the supply of sediment and the capacity for sediment to be transported, it may have a direct connection to many anthropogenic disturbances and is a useful synthetic measure of sediment dynamics.

RBS is estimated following Kaufmann et al. (2008, 2009) as the log ratio of observed geometric mean bed surface particle diameter (from 155 substrate particles classified by field crews) and the "critical" or mobile particle diameter estimated for a stream flow at bankfull flood conditions. To estimate the critical bed particle diameter in a natural stream, we calculate shear stress using bankfull flow depth, slope, channel shape, and channel roughness, such as the mitigating role of large woody debris, and channel complexity (Simons and Senturk 1977; Kaufmann et al. 2008). By using the modeled critical particle diameter, RBS is in effect based on the expected conditions at each site in Glacier. An RBS value of 1.0 (which is equal to 0 when it is logged as in (Table 20) indicates that the mean bed surface particle size in a site is mobile during its bankfull flow.

3.4.6.2 Assessment points

Relevant ecological thresholds are available for all three of the more general substrate metrics. Bryce et al. (2010) defines habitat suitability thresholds for fines and sand using bull trout and cutthroat trout and a suite of eight macroinvertebrate taxa that occur in the park. The assessment points are defined as "optimum tolerance" percent sediment cover thresholds that protect and maintain self-sustaining populations of these native sediment-sensitive aquatic species. We use the values for macroinvertebrates as our assessment points as they are more conservative than the salmonid version and likely have wider applicability in the park. The best available threshold for embeddedness is from USFWS (1998); it sets the maximum percentage of larger substrates surrounded by sands and fines (embeddedness) at 20% in properly functioning bull trout rearing habitat.

Kaufmann et al. ("Physical habitat in conterminous US streams and rivers, Part 1," 2022, "Physical habitat in conterminous US streams and rivers, Part 2," 2022) report that across 4,193 US stream sites, typical logged RBS range from −1.5 to +0.5, with values of −4.5 for very mobile streambeds composed of fine sediments to +3.5 for stable beds, usually consisting of coarse or large particles or

bedrock. Values of RBS for a given stream type and geoclimatic setting can be either too low (finer, more unstable streambeds) or too high (coarser, more stable streambeds) compared with expected under minor disturbance or reference conditions. We define assessment points on both sides of the RBS gradient (Figure 53 shows this graphically) from 75 ecoregional sites. In this report we focus on the two boundary values defining non-reference or major stress fining and non-reference armoring. Fining may result either from high inputs of fine sediments (i.e., when the supply of sediments from the landscape exceeds the ability of the stream to move them downstream) or increases in flood magnitude or frequency. Low RBS resulting from hydrologic alteration can be a precursor to channel incision. In contrast, armoring is typified by hard, armored streambeds, such as those often found below large dams where fine sediment flows are interrupted or within channels where banks are highly altered (e.g., paved or lined with riprap).

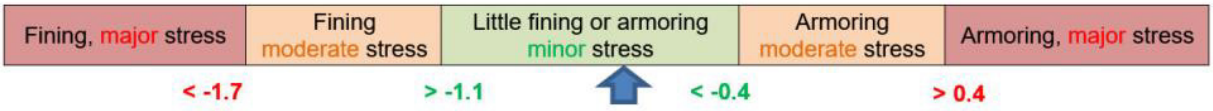


Figure 53. Arrangement of major and minor assessment points for relative bed stability. Fining happens when excessive fine sediment loading is present, and armoring is the presence of too many coarse sediments. Unlabeled orange regions are intermediate between reference and non-reference states. Blue arrow indicates approximate location of the median Glacier National Park relative bed stability value in 2007–2009.

3.4.6.3 Parkwide 2007–2009 baseline results

Schweiger et al. (2018) found generally low levels of fines and sands relative to the assessment point thresholds for salmonids and invertebrates across the baseline population of streams in the park. These results reflect the dominant larger substrate types in most Glacier streams and the largely balanced sediment dynamics typical of unmodified channels in the park. However, baseline median embeddedness in Glacier was slightly higher than the percent fines and sand metrics would suggest and exceeded the assessment point for bull trout habitat quality in a higher percentage of the park's streams than was expected. Embeddedness is not collected from non-wadeable sites due to safety concerns. The thalweg of larger rivers would generally have little to no embedded substrates, so there may have been some unaccounted bias towards smaller streams with higher embeddedness in baseline values. Results for the more nuanced RBS metric also suggested most streams in the park were in a reference range, neither fining nor armoring. A few fining sites were likely due to nearby sediment input from roads. A few others streams with more fines than expected but in a fully wilderness context with no known human disturbances were more difficult to interpret. However, overall, except for front country streams with clear anthropogenic issues (i.e., Divide Creek designated under the Clean Water Act as sediment impaired), fine sediment issues were not a concern in the baseline data.

3.4.6.4 Trend

There were no significant or marginally significant trends in any sediment indicator at any sentinel site (Table 22, Figure 54, Figure 55). The directions of the slopes varied with site and indicator, but

overall, sediment as estimated by these indicators at these sites was stable over our period of record. However, some trends approached a marginal significance level. For example, there were decreasing level of fines at Reynolds Creek. There is also a suggestion of some increased fining (negative slope across time) at Swiftcurrent Creek, perhaps related to the beaver dam in place between 2014 and 2016 and how this affected sediment in channels. These patterns need to be confirmed with more data.

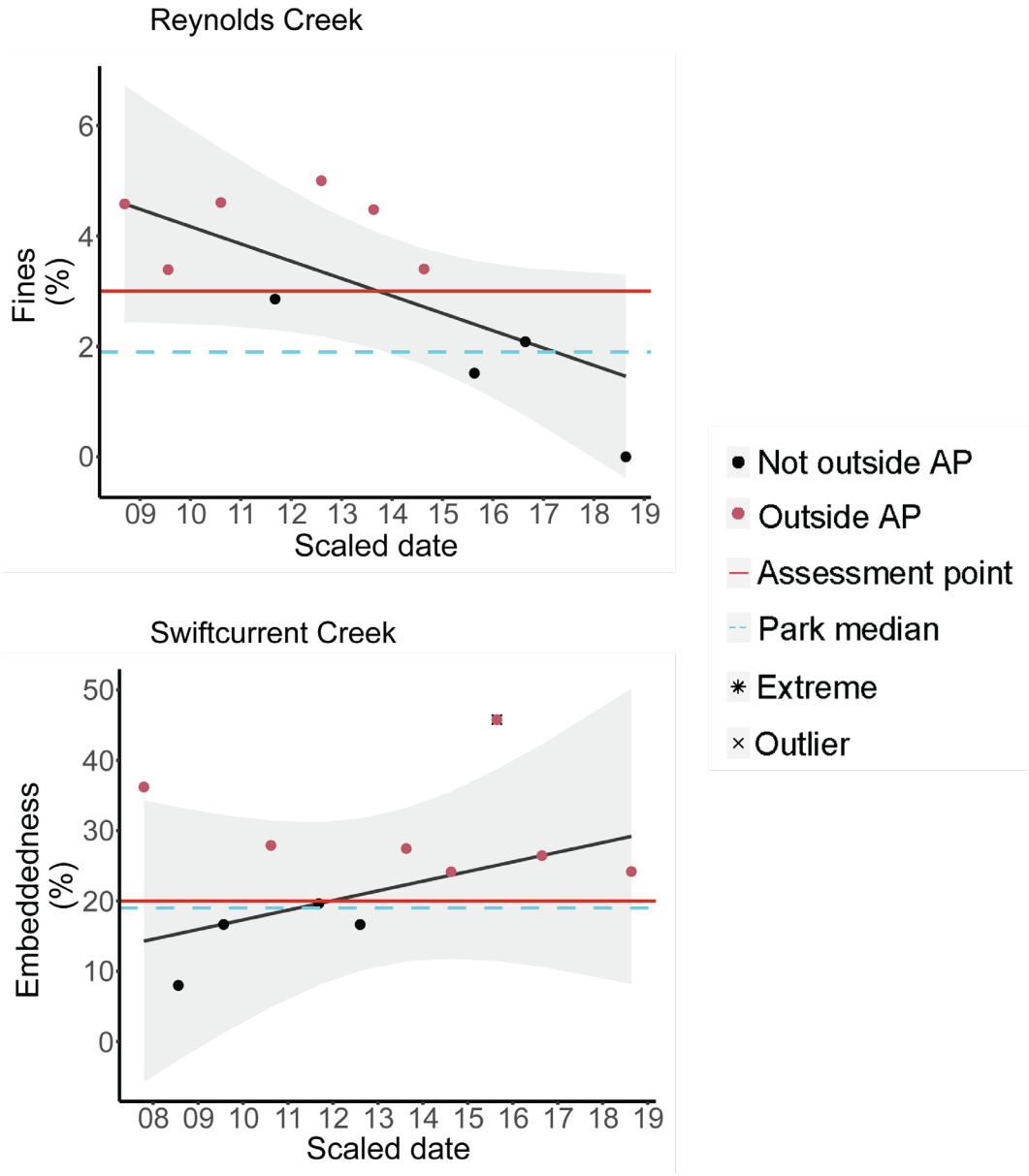


Figure 54. Examples of indicators of sediment in Glacier National Park from 2007 to 2019. (Top) Percent fines at Reynolds Creek, (bottom) embeddedness at Swiftcurrent Creek. As available, red horizontal lines show the non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (no significant trends for either metric at these two sites) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

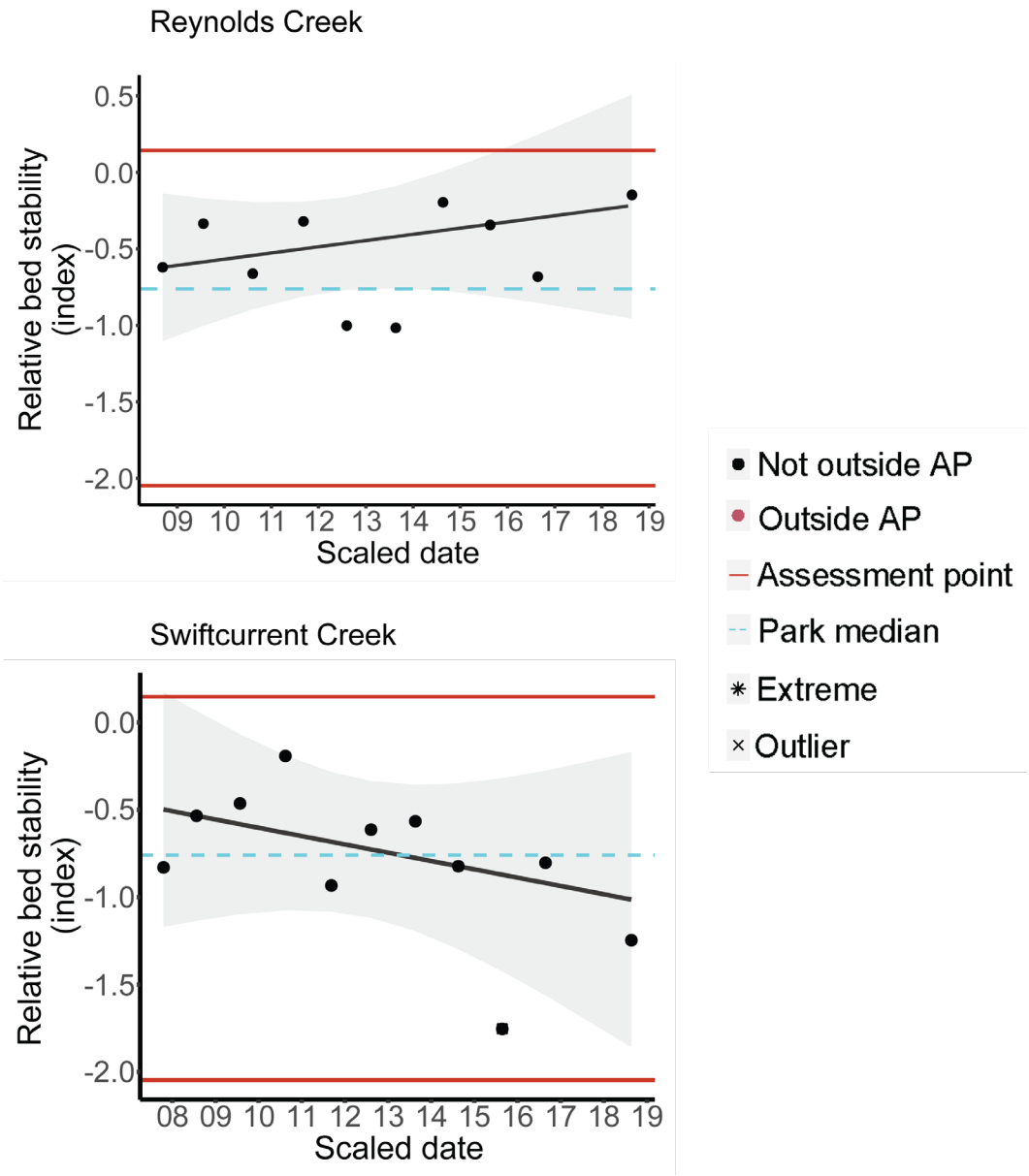


Figure 55. Relative bed stability in Glacier National Park from 2007 to 2019. (Top) Reynolds Creek, and (bottom) Swiftcurrent Creek. As available, red horizontal lines show the non-reference AP and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (neither show significant trends) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

3.4.6.5 Condition

Several sediment results were outside a reference range suggested by our assessment point (Table 20, Figures 54–55). The two sites where embeddedness was measured that have or could have bull trout populations (Ole and Swiftcurrent Creeks, respectively) had at least one value outside the bull trout habitat suitability threshold for embeddedness. Because this assessment point is fairly conservative and embeddedness is highly naturally variable in time and space, we do not see these results as problematic. A higher percentage of the fines (20% to 60%) and sand and fines (0% to 63.6%) at our

sentinel sites during our period of record were outside the ecological threshold for macroinvertebrate habitat suitability (Table 20). This is a rigorous threshold derived using a large dataset that includes sites in and around Glacier and is likely a good approximation of a causal or structural relationship between macroinvertebrate habitat quality and smaller sediments in the park's streams.

While the percentages of fines and sands outside the assessment point may warrant some concern, most sites had relative bed stability in a reference range suggested by ecoregional assessment points (Table 20). All sites except the Middle Fork Flathead (see paragraph below) were on the fining side (negative log RBS values), but above the lower assessment point of −2.047. These assessment points may not have the ecological basis that the percent fines and sand + fines assessment points do from Bryce et al. (2010). The ecoregional values we estimated for RBS from reference site data are similar to those in Stoddard et al. (2005), also for the Northern/Canadian Rockies ecoregion, that were derived from regional scientists based on their expert opinion—this does not make them regulatory or ecological, but the overlap is useful.

The Middle Fork Flathead RBS values stand out as being more on the armoring side. We only have two data points for this site in the dataset for this report, but both were outside the assessment point on the armoring or large sediment side of the RBS gradient. There are also lower levels of fines and sand + fines at this site. While there is some human land use at or near this site (see Figure 8), including limited and localized rip rap, which elevates RBS, we think the higher RBS is due to a meaningful portion of the sample reach that has natural and exposed bedrock as the stream bottom. Bedrock is, by definition, armored, but this is a natural part of this river. Some glacial meltwater streams with coarse beds are not able to transport the (cobble) size of bed particles they inherit from a melting glacier. This means that they also cannot make scour pools, so, like a stream choked with mobile bedload, they have plane-bed channels without marked pool-riffle structure (P. Kaufmann, US EPA hydrologist, pers. comm.).

3.4.6.6 Discussion and management applications

There are no relevant state or federal regulatory criteria for our indicators of sediment. The ecological threshold assessment points we apply likewise have no regulatory heft, but they are well researched; therefore, higher than reference level small sediment percentages at several of our sites may be of some concern. RBS is a more rigorous sediment metric than the more general percent fines or sand + fines, but its ecoregional assessment point is less diagnostic than the ecological thresholds in Bryce et al. (2010). Additional data are needed to resolve this issue.

We can bring to bear on this sediment issue related indicators from other areas of the SEI protocol—therefore we include diverse ways of monitoring condition in our approach. We do not see any meaningful issues with total suspended sediment in our water chemistry data at any site, including the Swiftcurrent Creek or Reynolds Creek sites (where we see more small sediments on the channel bed). Sediment specific bioindicators for macroinvertebrates suggest some negative response to fines at Swiftcurrent Creek and the North Fork Flathead—Border site. The other sites were in a reference condition over the period of record. For diatoms, the sediment increaser metric only had one result over the assessment point during our period of record, suggesting a small diatom response to fine sediments. Diatom sediment increasers at the North Fork Flathead—Border site and Reynolds Creek

did approach the assessment point in one additional event, suggesting that, at these sites, diatoms were slightly more reflective of increased sediments (but see Weilhoefer and Pan 2022). Overall, there was not a strong biological response to fine sediments, suggesting that the higher levels of measured sediments as part of our habitat protocol did not translate to a meaningful impact to a site's biology.

In summary, our data paint a complex picture regarding sediments at these sites. As we discuss above in the water chemistry section, the processes driving sediment loads in streams and rivers are complex (Meybeck et al. 2003). However, climate change has impacted and will continue to impact precipitation and runoff as well as vegetative cover in Glacier National Park, which influences sediment transport processes into streams and rivers (Chaplot 2007; Ficklin et al. 2013). The high levels of small sediments in many of our sites and suggestions of increased fining in some sites warrants some concern. The lack of an actual negative response in macroinvertebrates and diatoms is encouraging.

3.4.7 Floodplain interaction

3.4.7.1 Background and relevancy

Following Stoddard et al. (2005), Madej (2001), and Kaufmann et al. (1999), we use four indicators of the interactions between stream channels and their floodplains: (1) sinuosity, 2) incision, 3) bankfull width to depth ratio, and 4) bankfull width to wetted width ratio.

Bankfull width to depth ratio and the bankfull width to wetted width ratio are integrators of channel response to changes in sediment supply and transport, with important relationships to over bank flooding, channel shading, hyporheic flow (Kasahara and Wondzell 2003), stream temperature (White et al. 1987), and deep water habitat refugia for fish (Montana DEQ 2011). Bankfull width to wetted width estimates the potential for inundation from floods—sites with wide lateral bars (and thus higher ratios) reduce the likelihood of floods interacting with the floodplain. In most stream types in more intact mountain watershed systems like Glacier National Park, channels are characterized by lower width to depth ratios (i.e., channels are not a "mile wide and an inch deep" like they can naturally be in lower elevation areas). Likewise, except in larger valley bottom rivers, most streams in Glacier have less extensive lateral bars separating wetted and bankfull width (Rosgen 1994).

Channel sinuosity of a stream or river reach is the ratio of channel length as the water flows divided by the straight-line distance between the two ends of a defined reach. The greater the sinuosity the slower water flows through a valley and the more chance it must interact with the terrestrial landscape. Decreased sinuosity tends to increase channel slope and can lead to lower relative bed stability, greater sediment transport, increased bank instability, channel downcutting, and downstream flooding. Stream incision occurs when the channel bed has dropped in elevation and the stream is disconnected from its floodplain. This leaves bankfull stage at a lower elevation than the top of the stream bank. It is usually caused by an increase in the erosive ability of a river, a decrease in the bedload sediment supply, or when the stream bottom and banks are destabilized by disturbances—either natural or human caused (Poff et al. 1997; Simon and Rinaldi 2006).

3.4.7.2 Assessment points

Bankfull width to wetted width ratio and incision increase as condition in a stream deteriorates, whereas bankfull width to depth ratio and sinuosity decrease as condition in a stream deteriorates. There are no management, regulatory, or known ecological assessment points for the four floodplain interaction indicators, so we estimate assessment points from 75 ecoregional reference sites.

3.4.7.3 Parkwide 2007–2009 baseline results

Around 17% of the park's streams in 2007–2009 had sinuosity in a non-reference condition based on the ecoregional assessment point. However, while a few sites with lower sinuosity are due to adjacent road constraints, most of these simpler straighter channels result from natural confining features like cliffs or bedrock. In general, streams in Glacier National Park have not been subjected to extensive human disturbances that have reduced sinuosity. Likewise, incision was largely in a reference condition, with few human drivers. Bankfull width to depth ratio and bankfull width to wetted width were also largely in a reference state, with little indication of meaningful issues from human caused imbalances in sediment and discharge dynamics.

3.4.7.4 Trend

There were no significant or marginally significant trends in bankfull width to depth ratio or incision at any sentinel site (Table 21, Figure 56). The directions of slopes varied with site and indicator, but overall, these aspects of stream–floodplain interactions were stable over our period of record. There were two significant or marginally significant trends in other stream–floodplain interaction indicators (Table 21, Figure 56). Bankfull width to wetted width at the North Fork Flathead—Border marginally decreased, and sinuosity at our Swiftcurrent Creek site significantly increased.

Table 21. Select indicators of floodplain interaction at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Bankfull width / wetted width and incision increase as condition in a stream deteriorates and values above or equal to the assessment point (AP) are in a non-reference condition. Bankfull width / depth and sinuosity decrease as condition in a stream deteriorates and values below or equal to the AP are in a non-reference condition. Assessment point types were all ecoregional. We use the 5th and 95th percentile value from 75 ecoregional reference sites as AP for increasing and decreasing indicators, respectively. The percent of results outside the AP were all zero.

Variable	Site	Trend	Trend (p)	AP	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Bankfull width / depth (ratio)	North Fork Flathead—Glacier Rim	– [a]	– [a]	5.1783	8/15/2008	8/20/2019	3	2.411	119.906	119.380	3.252	115.897	122.337
	North Fork Flathead—Border	−1.5707	0.646	5.1783	7/11/2008	8/22/2019	10	2.411	62.747	68.930	29.333	44.254	146.242
	Swiftcurrent Creek	0.0087	0.987	5.1783	10/18/2007	8/22/2018	11	2.411	16.542	16.352	3.582	7.914	21.384
	Reynolds Creek	−0.6173	0.327	5.1783	9/11/2008	8/21/2018	10	2.411	10.606	12.049	5.755	6.971	26.882
	Ole Creek	0.6822	0.376	5.1783	8/27/2009	8/23/2018	9	2.411	10.382	11.841	3.964	6.426	18.929
	Middle Fork Flathead	– [a]	– [a]	5.1783	8/28/2018	8/19/2019	2	2.411	59.838	59.838	8.373	53.917	65.759
Bankfull width / wetted width (ratio)	North Fork Flathead—Glacier Rim	– [a]	– [a]	3.046	8/15/2008	8/20/2019	3	1.139	1.323	1.345	0.043	1.318	1.394
	North Fork Flathead—Border	−0.0833	0.109	3.046	7/11/2008	8/22/2019	10	1.139	1.314	1.515	0.444	1.148	2.604
	Swiftcurrent Creek	−0.0134	0.409	3.046	10/18/2007	8/22/2018	11	1.139	1.235	1.232	0.104	1.088	1.398

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] There is insufficient variation in sinuosity at North Fork Flathead—Border for quantile regression model estimation.

[c] Trend is significant ($p \leq 0.10$).

Table 21 (continued). Select indicators of floodplain interaction at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Bankfull width / wetted width and incision increase as condition in a stream deteriorates and values above or equal to the assessment point (AP) are in a non-reference condition. Bankfull width / depth and sinuosity decrease as condition in a stream deteriorates and values below or equal to the AP are in a non-reference condition. Assessment point types were all ecoregional. We use the 5th and 95th percentile value from 75 ecoregional reference sites as AP for increasing and decreasing indicators, respectively. The percent of results outside the AP were all zero.

Variable	Site	Trend	Trend (p)	AP	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Bankfull width / wetted width (ratio) (continued)	Reynolds Creek	−0.0198	0.643	3.046	9/11/2008	8/21/2018	10	1.139	1.343	1.350	0.233	0.986	1.688
	Ole Creek	−0.0235	0.728	3.046	8/27/2009	8/23/2018	9	1.139	1.501	1.507	0.297	1.069	1.952
	Middle Fork Flathead	– [a]	– [a]	3.046	8/28/2018	8/19/2019	2	1.139	1.294	1.294	0.086	1.233	1.354
Incision (m)	North Fork Flathead—Glacier Rim	– [a]	– [a]	3.018	8/15/2008	8/20/2019	3	0.470	0.000	0.000	0.000	0	0
	North Fork Flathead—Border	0.0000	1.000	3.018	7/11/2008	8/22/2019	10	0.470	0.000	0.110	0.269	0	0.659
	Swiftcurrent Creek	−0.0319	0.744	3.018	10/18/2007	8/22/2018	11	0.470	0.481	0.496	0.418	0	1.252
	Reynolds Creek	−0.0910	0.162	3.018	9/11/2008	8/21/2018	10	0.470	0.010	0.180	0.325	0	0.832
	Ole Creek	−0.0036	0.968	3.018	8/27/2009	8/23/2018	9	0.470	0.446	0.631	0.759	0	2.609
	Middle Fork Flathead	– [a]	– [a]	3.018	8/28/2018	8/19/2019	2	0.470	0.000	0.000	0.000	0	0

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] There is insufficient variation in sinuosity at North Fork Flathead—Border for quantile regression model estimation.

[c] Trend is significant ($p \leq 0.10$).

Table 21 (continued). Select indicators of floodplain interaction at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Bankfull width / wetted width and incision increase as condition in a stream deteriorates and values above or equal to the assessment point (AP) are in a non-reference condition. Bankfull width / depth and sinuosity decrease as condition in a stream deteriorates and values below or equal to the AP are in a non-reference condition. Assessment point types were all ecoregional. We use the 5th and 95th percentile value from 75 ecoregional reference sites as AP for increasing and decreasing indicators, respectively. The percent of results outside the AP were all zero.

Variable	Site	Trend	Trend (p)	AP	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Sinuosity (index)	North Fork Flathead—Glacier Rim	– [a]	– [a]	1.0124	8/15/2008	8/20/2019	3	1.041	1.349	1.349	0.000	1.349	1.349
	North Fork Flathead—Border	N/A [b]	N/A [b]	1.0124	7/11/2008	8/22/2019	10	1.041	1.133	1.133	0.000	1.133	1.133
	Swiftcurrent Creek	0.0034	**0.047** [c]	1.0124	10/18/2007	8/22/2018	11	1.041	1.103	1.104	0.012	1.086	1.118
	Reynolds Creek	−0.0096	0.170	1.0124	9/11/2008	8/21/2018	10	1.041	1.133	1.135	0.054	1.085	1.245
	Ole Creek	−0.0010	0.866	1.0124	8/27/2009	8/23/2018	9	1.041	1.142	1.148	0.025	1.115	1.189
	Middle Fork Flathead	– [a]	– [a]	1.0124	8/28/2018	8/19/2019	2	1.041	1.124	1.124	0.000	1.124	1.124

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] There is insufficient variation in sinuosity at North Fork Flathead—Border for quantile regression model estimation.

[c] Trend is significant ($p \leq 0.10$).

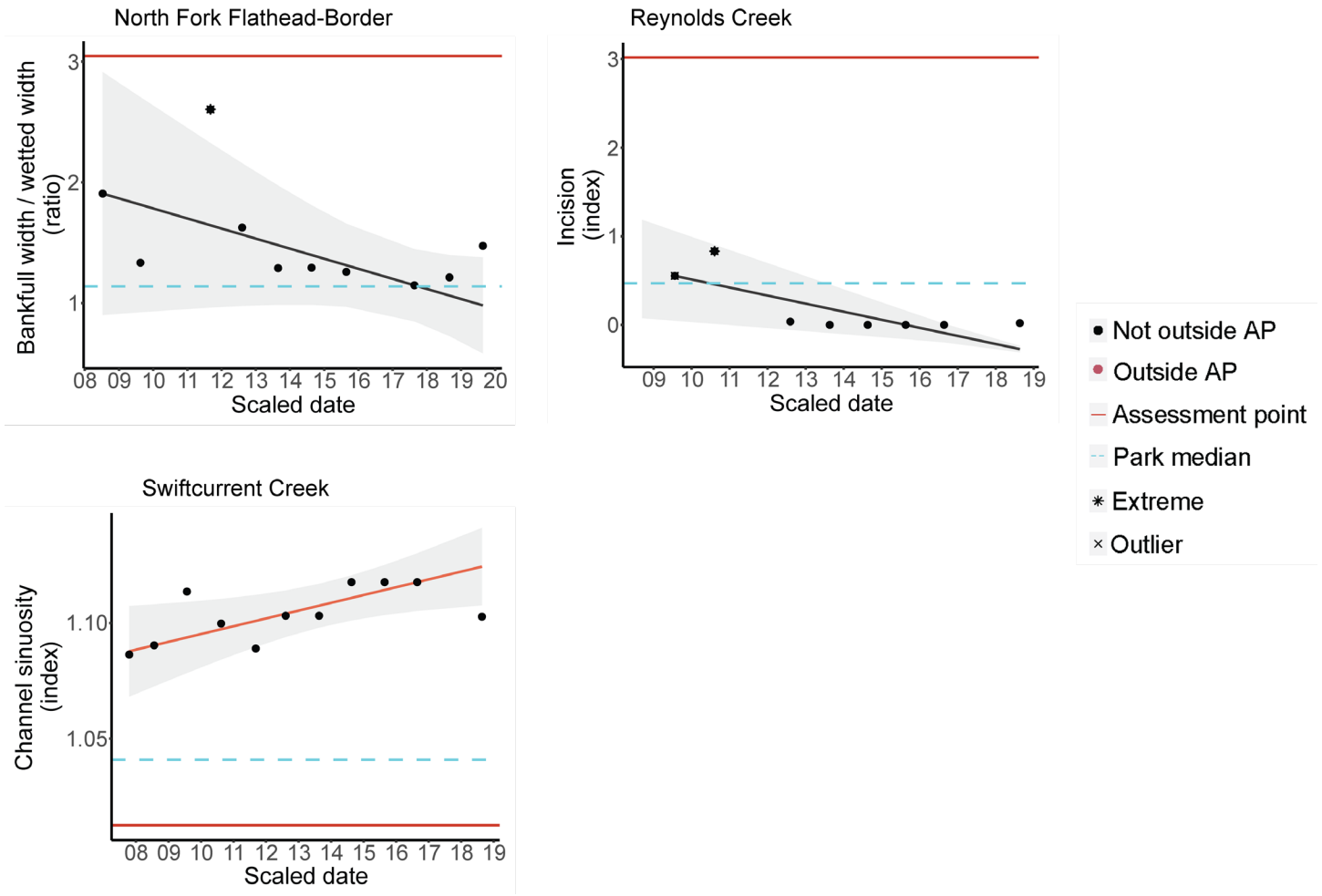


Figure 56. Examples of indicators of floodplain interaction in Glacier National Park from 2007 to 2019. (Top left) bankfull width to wetted width ratio at North Fork Flathead—Border, (top right) incision at Reynolds Creek, and (bottom left) sinuosity at Swiftcurrent Creek. As available, red horizontal lines show the non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (significant trend for channel sinuosity at Swiftcurrent Creek) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

3.4.7.5 Condition

No results for any of the stream–floodplain interaction indicators at our sentinel sites were outside the ecoregional assessment point (Table 21, Figure 56). This suggests that, for these indicators at our sites, stream channels had as good or better interactions with their floodplains over our period of record than the best sites in the surrounding ecoregions. Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead had floodplain interactions similar to other site results and in keeping with expectation for these larger rivers.

3.4.7.6 Discussion and management applications

Bankfull channel width is a fundamental aspect of streams and a key parameter of interest for many applications in hydrology, channel morphology, and stream ecology (Faustini et al. 2009). The decreasing ratio in bankfull width to wetted width at the North Fork Flathead—Border suggests that bankfull has decreased relative to wetted width (although wetted width might have also changed, and this needs to be further explored). The net change is small (from a ratio of about 2 to 1.5) but in general, there was less distance from the river's wetted edge to its bankfull extent across the years of our sampling. Interestingly, Faustini et al. (2009) show that the dominant response of channel width to human disturbance, particularly in mountain streams, is a reduction in bankfull width in streams with greater disturbance. While there is very little human disturbance in the North Fork watershed, this pattern may warrant some additional work. Exploratory models do suggest some relationships with water balance variables, in particular evapotranspiration, which might also suggest a climate link.

Incision is an important indicator of human disturbance on sediment regimes. High levels of sinuosity are to be expected given the wilderness nature of our sentinel sites. Incision is a natural process and usually develops slowly over time in undisturbed systems. Large human disturbances, like catastrophic "natural" events, greatly compress time scales for incision and related processes by creating enormous imbalances between upstream sediment delivery and available transporting power (Simon and Rinaldi 2006). Streams in Glacier National Park do experience large natural disturbances, and these can cause some channel incision. Incision at Ole Creek approached the assessment point in sample events that followed heavy spring runoff and a reworking of the channel, with large deposits of substrate into lateral bars and the creation of a complex bank structure (for example, Figure 57 below). This often formed partially incised channels, disconnected from the main floodplain; but these incised channels were gone later in the same year or the next. High energy systems like Ole Creek have active bedform that naturally creates such structure and heterogeneity and we do not view this as any cause for concern. Similar processes happened on the bigger river sites, but with generally larger channel dimensions, the relative incision was less. Note also that the ecoregional assessment point is quite large (~3 m). The overall median incision value in our sentinel site data was 0.76 m, and the unbiased parkwide incision in 2007–2009 was 0.47 m. These show that incision in Glacier is notably less than in ecoregional reference sites.



Figure 57. Rocky Mountain Network stream ecological integrity field crew collect data on riparian vegetation cover and composition at Ole Creek in Glacier National Park, 2010. The image also shows the typical complex channel and bank structure at Ole Creek (NPS / BILLY SCHWEIGER).

Finally, the increased sinuosity at Swiftcurrent Creek is unusual as this sample reach is constrained by a large hillslope to the south and the campground to the north. As is apparent on imagery (see Figure 37), most of the sampled reach is quite straight. The increase we see in our sinuosity metric corresponds with the beaver dam and may be because the channel downstream of the dam became dewatered, with some small-scale sinuous braiding, and this was captured in our field data. We do not interpret this as any sort of human disturbance impact on channel sinuosity, which usually straightens a channel.

3.4.8 Riparian vegetation cover and invasive/exotic plants

3.4.8.1 Background and relevancy

Following Stoddard et al. (2005), Ringold et al. (2008), and Kaufman (2022a, b), we used four metrics describing riparian vegetation in Glacier National Park: 1) total areal cover of riparian vegetation in canopy plus mid-level plus ground layers, 2) total areal cover of riparian woody vegetation in canopy plus mid-level plus ground layers, 3) total cover of any feature (vegetation, hillslopes, etc.) at bankfull, and 4) invasive/exotic plant occurrence (Table 22).

Table 22. Select indicators of riparian vegetation at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. AP types include ecological threshold (THR), and ecoregional (ERG). The three riparian vegetation cover indicators decrease as condition in a stream deteriorates, and values below or equal to the AP are in a non-reference condition. Invasive/exotics increase as condition in a stream deteriorates, and values above or equal to the AP are in a non-reference condition. We use the 5th value from 75 ecoregional reference sites as AP for the decreasing indicators. The AP for invasive/exotics is a Glacier National Park resource management directed value.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Riparian vegetation cover (all) (proportion)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	0.4864	ERG	8/15/2008	8/20/2019	3	1.100	1.420	1.458	0.593	0.885	2.069
	North Fork Flathead—Border	−0.0299	0.520	0.0	0.4864	ERG	7/11/2008	8/22/2019	10	1.100	1.141	1.150	0.322	0.689	1.686
	Swiftcurrent Creek	0.0031	0.948	0.0	0.4864	ERG	10/18/2007	8/22/2018	11	1.100	1.215	1.274	0.356	0.877	2.077
	Reynolds Creek	0.0056	0.735	10.0	0.4864	ERG	9/11/2008	8/21/2018	10	1.100	0.608	0.615	0.094	0.464	0.803
	Ole Creek	0.0340	0.456	0.0	0.4864	ERG	8/27/2009	8/23/2018	9	1.100	0.781	0.844	0.227	0.59	1.369
	Middle Fork Flathead	– [a]	– [a]	0.0	0.4864	ERG	8/28/2018	8/19/2019	2	1.100	1.976	1.976	0.333	1.741	2.211

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] Trend is significant ($p \leq 0.10$).

[d] We use the park-scale mean instead of the median for the Invasive Plant Index.

[e] N/A No invasive/exotics were observed at Reynolds Creek for quantile regression model estimation.

Table 22 (continued). Select indicators of riparian vegetation at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. AP types include ecological threshold (THR), and ecoregional (ERG). The three riparian vegetation cover indicators decrease as condition in a stream deteriorates, and values below or equal to the AP are in a non-reference condition. Invasive/exotics increase as condition in a stream deteriorates, and values above or equal to the AP are in a non-reference condition. We use the 5th value from 75 ecoregional reference sites as AP for the decreasing indicators. The AP for invasive/exotics is a Glacier National Park resource management directed value.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Riparian vegetation cover (woody) (proportion)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	0.1807	ERG	8/15/2008	8/20/2019	3	0.680	1.111	1.084	0.497	0.575	1.567
	North Fork Flathead—Border	−0.0226	0.593	0.0	0.1807	ERG	7/11/2008	8/22/2019	10	0.680	0.781	0.803	0.256	0.46	1.186
	Swiftcurrent Creek	0.0047	0.897	0.0	0.1807	ERG	10/18/2007	8/22/2018	11	0.680	0.802	0.900	0.299	0.59	1.639
	Reynolds Creek	0.0006	0.960	0.0	0.1807	ERG	9/11/2008	8/21/2018	10	0.680	0.290	0.288	0.059	0.195	0.392
	Ole Creek	0.0301	0.434	0.0	0.1807	ERG	8/27/2009	8/23/2018	9	0.680	0.545	0.606	0.201	0.409	1.076
	Middle Fork Flathead	– [a]	– [a]	0.0	0.1807	ERG	8/28/2018	8/19/2019	2	0.680	1.506	1.506	0.119	1.422	1.59

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] Trend is significant ($p \leq 0.10$).

[d] We use the park-scale mean instead of the median for the Invasive Plant Index.

[e] N/A No invasive/exotics were observed at Reynolds Creek for quantile regression model estimation.

Table 22 (continued). Select indicators of riparian vegetation at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. AP types include ecological threshold (THR), and ecoregional (ERG). The three riparian vegetation cover indicators decrease as condition in a stream deteriorates, and values below or equal to the AP are in a non-reference condition. Invasive/exotics increase as condition in a stream deteriorates, and values above or equal to the AP are in a non-reference condition. We use the 5th value from 75 ecoregional reference sites as AP for the decreasing indicators. The AP for invasive/exotics is a Glacier National Park resource management directed value.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Total bank cover (%)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	5.5882	ERG	8/15/2008	8/20/2019	3	58.900	11.631	13.102	3.267	10.829	16.845
	North Fork Flathead—Border	0.0435	0.846	**40.0** [b]	5.5882	ERG	7/11/2008	8/22/2019	10	58.900	5.735	5.039	1.490	2.674	6.684
	Swiftcurrent Creek	4.8408	**0.009** [c]	9.1	5.5882	ERG	10/18/2007	8/22/2018	11	58.900	44.818	33.784	21.106	4.813	59.893
	Reynolds Creek	4.0145	0.154	20.0	5.5882	ERG	9/11/2008	8/21/2018	10	58.900	24.465	22.825	15.820	3.476	43.873
	Ole Creek	2.5323	0.477	0.0	5.5882	ERG	8/27/2009	8/23/2018	9	58.900	41.486	39.256	14.425	17.112	58.263
	Middle Fork Flathead	– [a]	– [a]	0.0	5.5882	ERG	8/28/2018	8/19/2019	2	58.900	9.024	9.024	2.174	7.487	10.561

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] Trend is significant ($p \leq 0.10$).

[d] We use the park-scale mean instead of the median for the Invasive Plant Index.

[e] N/A No invasive/exotics were observed at Reynolds Creek for quantile regression model estimation.

Table 22 (continued). Select indicators of riparian vegetation at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside is the percentage of results above (and/or below) the given assessment point(s) (AP). Any Percent outside value above or equal to 25% is in bold. AP types include ecological threshold (THR), and ecoregional (ERG). The three riparian vegetation cover indicators decrease as condition in a stream deteriorates, and values below or equal to the AP are in a non-reference condition. Invasive/exotics increase as condition in a stream deteriorates, and values above or equal to the AP are in a non-reference condition. We use the 5th value from 75 ecoregional reference sites as AP for the decreasing indicators. The AP for invasive/exotics is a Glacier National Park resource management directed value.

Variable	Site	Trend	Trend (p)	Percent outside AP	AP	AP type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Invasive plants (index)	North Fork Flathead—Glacier Rim	– [a]	– [a]	**100.0** [b]	0.000	THR	8/15/2008	8/20/2019	3	0.230 [d]	0.625	0.633	0.455	0.182	1.091
	North Fork Flathead—Border	0.0355	0.647	**80.0** [b]	0.000	THR	7/11/2008	8/22/2019	10	0.230 [d]	0.668	0.772	0.629	0	2.091
	Swiftcurrent Creek	−0.0131	0.719	**90.9** [b]	0.000	THR	10/18/2007	8/22/2018	11	0.230 [d]	0.273	0.297	0.191	0	0.615
	Reynolds Creek	N/A [e]	N/A [e]	0.0	0.000	THR	9/11/2008	8/21/2018	10	0.230 [d]	0.000	0.000	0.000	0	0
	Ole Creek	0.0275	0.683	100.0	0.000	THR	8/27/2009	8/23/2018	9	0.230 [d]	0.727	0.727	0.360	0.091	1.455
	Middle Fork Flathead	– [a]	– [a]	100.0	0.000	THR	8/28/2018	8/19/2019	2	0.230 [d]	0.642	0.642	0.082	0.583	0.7

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Percent outside value ≥ 25%.

[c] Trend is significant ($p \leq 0.10$).

[d] We use the park-scale mean instead of the median for the Invasive Plant Index.

[e] N/A No invasive/exotics were observed at Reynolds Creek for quantile regression model estimation.

Riparian vegetation cover is collected in three canopy height strata, with values ranging from 0 to 3.0. This is expressed as a percentage; each layer may have 100% cover, and across three strata, the value can theoretically sum to 300% or 3.0 as a proportion. For canopy cover, we use data collected on the bank (versus mid-channel, as in Stoddard et al. 2005), as this is more appropriate for a dataset like ours that includes larger rivers that are naturally too wide for a mid-channel canopy. Canopy cover also includes all natural cover (not just vegetation) at 30 cm above the ground. This can be important in shade and thermal regulation in alpine and other naturally sparsely vegetated areas in the park (Vannote et al. 1980).

Invasive/exotic plants

Understanding and monitoring invasive/exotic taxa (Figure 58) in riparian systems is key for park resource management. Invasive/exotic riparian taxa have been associated with multiple negative effects (Ringold et al. 2008), including 1) alteration of fire, nutrient, or hydrologic regimes, 2) excess carbon sequestration, 3) changes to plant and animal distributions, and 4) repelling or sickening herbivores such as ungulates or livestock. Invasive/exotic plants are high visibility indicators for stream monitoring in the park and are a stand-alone vital sign for the Rocky Mountain Network (Britten et al. 2007). Monitoring the status and trends in invasive/exotic plants within riparian systems are of special interest for three reasons (Ringold et al. 2008). First, riparian areas may be more likely to be invaded than upland ecosystems (Brown and Peet 2003), given higher levels of both natural and anthropogenic disturbance (Hobbs and Huenneke 1992; Mack 2000; Nilsson and Svedmark 2002). Second, connectivity created by a stream network facilitates dispersal across a landscape (Merritt and Wohl 2002; Nilsson and Svedmark 2002). Third, many invasive/exotic species can alter stream ecosystem processes (i.e., Zavaleta 2000). Note that we also report on select aquatic invasive/exotic species in the bioassessment section below.



Figure 58. A patch of invasive/exotic oxeye daisy (Chrysanthemum leucanthemum; white flowers intermixed with willow seedlings) on the North Fork Flathead—Border, Glacier National Park, 2008. (NPS / BILLY SCHWEIGER)

Invasive/exotic species lists were developed with Glacier National Park staff in 2007 (see Table 23) and have been monitored annually since. (Note that reed canary grass, *Phalaris arundinacea*, was added to the list in 2010.) The list includes 16 taxa that are ecologically varied, relatively easy to identify, invasive, and ecologically or economically intrusive. Crews record occurrence at each of 22 riparian plots and express these data as a frequency adjusted by the total number of plots sampled (range of 0 to 1.0). An integrative metric called the Invasive Plant Index (IPI) is estimated as a sum of these frequencies across all taxa in a site. The IPI ranges from 0 to 16, with a score of 16 being the unlikely case of all species being found at all plots in a site.

Table 23. Mean baseline frequency (percentage of transects occupied) of 16 target invasive/exotic plant taxa at Glacier National Park during 2007–2009. Values are taken across all 87 sample events from 2007–2009. Montana Weed Category is from Montana Fish, Wildlife and Parks (2008). Category 1 is widespread noxious weeds and Category 2 is established new invaders. Reed canary grass was added after the baseline survey.

Taxa	Montana Weed Category	Mean site scale frequency (%)
Spotted knapweed (*Centaurea maculosa*)	1	0.084
Canada thistle (*Cirsium arvense*)	1	0.063
Oxeye daisy (*Chrysanthemum leucanthemum*)	1	0.049
St. Johnswort (*Hypericum perforatum*)	1	0.034
Meadow hawkweed complex (*Hieracium pratense, H. floribundum, H. piloselloides*)	2	0.033
Common tansy (*Tanacetum vulgare*)	1	0.009
Yellow toadflax (*Linaria vulgaris*)	1	0.008
Orange hawkweed (*H. aurantiacum*)	2	0.004
Dalmatian toadflax (*Linaria dalmatica)*	1	0.002
Leafy spurge (*Euphorbia esula*)	1	0
Houndstongue (*Cynoglossum officinale*)	1	0
Purple loosestrife (*Lythrum salicaria, L. virgatum*)	2	0
Tall buttercup (*Ranunculus acris*)	2	0
Tansy ragwort (*Senecio jacobea*)	2	0
Japanese knotweed (*Polygonium cuspidatum*)	2	0
Reed canary grass (*Phalaris arundinacea*)	N/A	N/A

Nine of the target taxa in Glacier National park are Category 1 on the state noxious weed list (MFWP 2008; USDA 2017). Category 1 taxa are generally widespread in the state and are well adapted to a wide range of site conditions. They are noxious, rendering land unfit or greatly limiting beneficial uses. The Montana Weed Management Plan lists widespread infestations of such Category 1 weeds as a third (or lower) priority for management, given their widespread nature and difficulty of control. Six additional taxa are Category 2 species that are rapidly spreading from current infestations. They are also considered noxious, rendering land unfit or greatly limiting beneficial uses. Their control is of a higher priority, given a greater possibility of success.

Caveats

There are some important caveats for our work with invasive/exotic plants. Our data are relatively simple and should not replace efforts by Glacier to monitor invasive vegetation. SEI sites were not targeted towards areas with higher probability of invasive species occurrence, and we likely miss local scale invasions. Finally, we do not conduct complete surveys of the vegetation in and around our sites because this would be cost and time prohibitive. Therefore, we do not have data on the important community context of invasive/exotic plants.

3.4.8.2 Assessment points

The three riparian cover indicators decrease as condition in a stream deteriorates, whereas invasive/exotic plants increase as condition in a stream deteriorates. There are no management, regulatory, or known ecological assessment points for the three cover metrics, so we estimate ecoregional assessment points from 75 ecoregional reference sites. For the invasive/exotic plants indicator, we use a Glacier National Park resource management defined/accepted assessment point of no occurrence of any of the 16 target taxa at a sentinel stream site (or an IPI score of 0). This assessment point originated with the assumption that a reference state is no exotic or nonnative species at all at a sentinel stream site in the park and is in keeping with the NPS approach to managing invasive/exotic plants (NPS 2006b; see also Ringold et al. 2008). Park management recognized that this is likely not a realistic goal, but felt it best reflected the purpose of a wilderness park like Glacier. We also recognize that an assessment of disturbance, alteration, or impairment relative to pristine condition is valuable in itself and should be separate from an assessment of the degree of alteration that is acceptable from an ecological, social, or economic perspective.

3.4.8.3 Parkwide 2007–2009 baseline results

All vegetation cover, woody vegetation cover, and total bank cover in the park was generally more extensive than at reference sites in the ecoregions represented in the park, with little widespread evidence of impact from human disturbance reducing cover. A few sites did have some disturbed vegetation (i.e., Divide Creek on the park's east side) where visitor services or other human driven disturbances impact cover, but this was rare, parkwide.

Interestingly, a large number of sites with invasive/exotic species in the survey suggested a strong and intuitive positive relationship between human disturbance and invasion by nonnative plants among sites. Invasive plants also increased with the density of stream crossings by roads within each site's watershed. These relationships are no longer present when observations are spread across a decade in the smaller sample of sentinel sites. This suggests variability in the controls on invasion is spatial, not temporal—this is an important result from our work.

Eight target taxa were observed at least once across all sentinel site events (Table 24). Spotted knapweed and oxeye daisy were the most recorded taxa, occurring in 57% and 43% of events across sites. The integrative IPI metric suggested that almost 40% of the park's streams in 2007–2009 did not meet the assessment point of zero invasive/exotics. We do not estimate species specific trend models, but Canada thistle, oxeye daisy and spotted knapweed were the most common species across all sentinel sites, with St. Johnswort and the meadow hawkweed complex abundant at Ole Creek. While we hesitate to make comparisons across sites without statistical analyses, the North Fork of the Flathead River had the highest levels of invasive/exotic species, with Ole Creek also having episodic higher levels of our target invasive plants.

Table 24. Invasive/exotic plant frequency across all subsample locations within each sample event, and the Invasive Plant Index that synthesizes scores across all target taxa at sentinel sites in Glacier National Park, 2007–2019. No invasive plants were found at Reynolds Creek and this site is omitted from calculations.

Site	Year	Reed canary grass (*Phalaris arundinacea*)	Common tansy (*Tanacetum vulgare*)	Canada thistle (*Cirsium arvense*)	Meadow hawkweed complex (*Hieracium pratense, H. floribundum, H. piloselloides*)	Oxeye daisy (*Chrysanthemum leucanthemum*)	St. Johnswort (*Hypericum perforatum*)	Spotted knapweed (*Centaurea maculosa*)	Yellow toadflax (*Linaria vulgaris*)	IPI Score
North Fork Flathead—Glacier Rim	2008	– [a]	0.09	– [a]	– [a]	0.36	– [a]	0.64	– [a]	1.091
	2018	– [a]	– [a]	– [a]	– [a]	0.18	– [a]	– [a]	– [a]	0.182
	2019	– [a]	– [a]	0.13	– [a]	0.13	– [a]	0.38	– [a]	0.625
North Fork Flathead—Glacier Rim Total	N/A [b]	N/A [b]	0.09	0.13	N/A [b]	0.22	N/A [b]	0.51	N/A b	0.633
North Fork Flathead—Border	2008	– [a]	– [a]	– [a]	– [a]	0.45	– [a]	– [a]	– [a]	0.455
	2009	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	0.000
	2011	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	0.000
	2012	0.36	– [a]	0.18	– [a]	0.82	– [a]	0.73	– [a]	2.091
	2013	– [a]	– [a]	0.18	0.09	0.09	– [a]	0.27	– [a]	0.636
	2014	0.18	– [a]	– [a]	– [a]	0.64	– [a]	0.64	– [a]	1.455
	2015	– [a]	– [a]	0.17	– [a]	0.50	– [a]	0.17	– [a]	0.833
	2017	– [a]	– [a]	– [a]	– [a]	0.64	– [a]	– [a]	– [a]	0.636
	2018	– [a]	– [a]	0.10	– [a]	0.50	– [a]	0.10	– [a]	0.700
	2019	– [a]	– [a]	– [a]	– [a]	0.45	– [a]	0.45	– [a]	0.909
North Fork Flathead—Border Total	N/A [b]	0.54	N/A [b]	0.16	0.09	0.51	N/A [b]	0.39	N/A b	0.772

[a] "–" (en-dash): Not detected.

[b] N/A: Not applicable.

Table 24 (continued). Invasive/exotic plant frequency across all subsample locations within each sample event, and the Invasive Plant Index that synthesizes scores across all target taxa at sentinel sites in Glacier National Park, 2007–2019. No invasive plants were found at Reynolds Creek and this site is omitted from calculations.

Site	Year	Reed canary grass (*Phalaris arundinacea*)	Common tansy (*Tanacetum vulgare*)	Canada thistle (*Cirsium arvense*)	Meadow hawkweed complex (*Hieracium pratense, H. floribundum, H. piloselloides*)	Oxeye daisy (*Chrysanthemum leucanthemum*)	St. Johnswort (*Hypericum perforatum*)	Spotted knapweed (*Centaurea maculosa*)	Yellow toadflax (*Linaria vulgaris*)	IPI Score
Swiftcurrent Creek	2007	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	0.000
	2008	– [a]	– [a]	0.09	– [a]	– [a]	– [a]	– [a]	– [a]	0.091
	2009	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	0.23	0.38	0.615
	2010	– [a]	– [a]	0.09	– [a]	– [a]	– [a]	0.18	0.18	0.455
	2011	– [a]	– [a]	0.09	– [a]	– [a]	– [a]	0.09	0.09	0.273
	2012	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	0.09	0.09	0.182
	2013	– [a]	– [a]	– [a]	– [a]	0.09	– [a]	0.09	0.18	0.364
	2014	– [a]	– [a]	– [a]	– [a]	0.09	– [a]	0.36	0.09	0.545
	2015	– [a]	– [a]	0.09	– [a]	– [a]	– [a]	0.09	0.18	0.364
	2016	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	0.10	0.10	0.200
	2018	– [a]	– [a]	– [a]	– [a]	– [a]	– [a]	0.18	– [a]	0.182
Swiftcurrent Creek Total	N/A [b]	N/A [b]	N/A [b]	0.09	N/A [b]	0.09	N/A [b]	0.16	0.16	0.297
Ole Creek	2009	– [a]	– [a]	– [a]	– [a]	– [a]	0.09	– [a]	– [a]	0.091
	2010	– [a]	– [a]	0.09	0.36	– [a]	0.18	– [a]	– [a]	0.636
	2011	0.18	– [a]	– [a]	0.36	– [a]	– [a]	0.09	– [a]	0.636
	2012	– [a]	– [a]	– [a]	0.73	– [a]	– [a]	– [a]	– [a]	0.727
	2013	– [a]	– [a]	– [a]	0.64	– [a]	0.09	0.09	– [a]	0.818

[a] "–" (en-dash): Not detected.

[b] N/A: Not applicable.

Table 24 (continued). Invasive/exotic plant frequency across all subsample locations within each sample event, and the Invasive Plant Index that synthesizes scores across all target taxa at sentinel sites in Glacier National Park, 2007–2019. No invasive plants were found at Reynolds Creek and this site is omitted from calculations.

Site	Year	Reed canary grass (*Phalaris arundinacea*)	Common tansy (*Tanacetum vulgare*)	Canada thistle (*Cirsium arvense*)	Meadow hawkweed complex (*Hieracium pratense, H. floribundum, H. piloselloides*)	Oxeye daisy (*Chrysanthemum leucanthemum*)	St. Johnswort (*Hypericum perforatum*)	Spotted knapweed (*Centaurea maculosa*)	Yellow toadflax (*Linaria vulgaris*)	IPI Score
Ole Creek (continued)	2014	– [a]	– [a]	0.09	0.82	– [a]	0.55	– [a]	– [a]	1.455
	2015	– [a]	– [a]	– [a]	0.55	– [a]	0.27	– [a]	– [a]	0.818
	2016	– [a]	– [a]	– [a]	– [a]	– [a]	0.50	– [a]	– [a]	0.500
	2018	– [a]	– [a]	– [a]	– [a]	– [a]	0.86	– [a]	– [a]	0.857
Ole Creek Total	N/A [b]	0.18	N/A [b]	0.09	0.58	N/A [b]	0.36	0.09	N/A [b]	0.727
Middle Fork Flathead	2018	– [a]	– [a]	0.30	– [a]	0.40	– [a]	– [a]	– [a]	0.700
	2019	– [a]	– [a]	– [a]	– [a]	0.33	0.17	0.08	– [a]	0.583
Middle Fork Flathead Total	N/A [b]	N/A [b]	N/A [b]	0.30	N/A [b]	0.37	0.17	0.08	N/A [b]	0.642
Mean event frequency	N/A [b]	0.72	0.09	0.13	0.51	0.38	0.34	0.25	0.16	0.592
Percent of events	N/A [b]	11.5	8.57	2.86	34.29	20.00	42.86	22.86	57.14	22.86

[a] "–" (en-dash): Not detected.

[b] N/A: Not applicable.

3.4.8.4 Trend

There were no significant or marginally significant trends in riparian vegetation cover of all types or in woody cover only (Table 22, Figure 59). Most vegetation cover slopes were positive but overall, our metrics of riparian vegetation were stable over our period of record at our sites. Of note, replicates across the years at most sites were fairly stable at the three wadeable sites (the top panel in Figure 59 provides a useful example) but variance was higher across years at the North Fork Flathead—Border site. An unknown portion of this variation is likely real but some of this is also methodological. It is more difficult to estimate cover values at large rivers as crews must visualize values on opposite banks of an often wide channel. At wadeable sites the riparian plot on both the left and right banks can typically be stood in or viewed easily adjacent to it for a more reliable estimation. Total bank cover increased significantly at Swiftcurrent Creek and approached marginal significance at Reynolds Creek (Table 22, Figure 59). The other two sites also had positive slopes in canopy cover. There were no significant or marginally significant trends in the Invasive Plant Index (Table 22, Figure 60). The direction of the slopes in the Invasive Plant Index varied, but in general the level of invasion at sentinel sites was stable over our period of record.

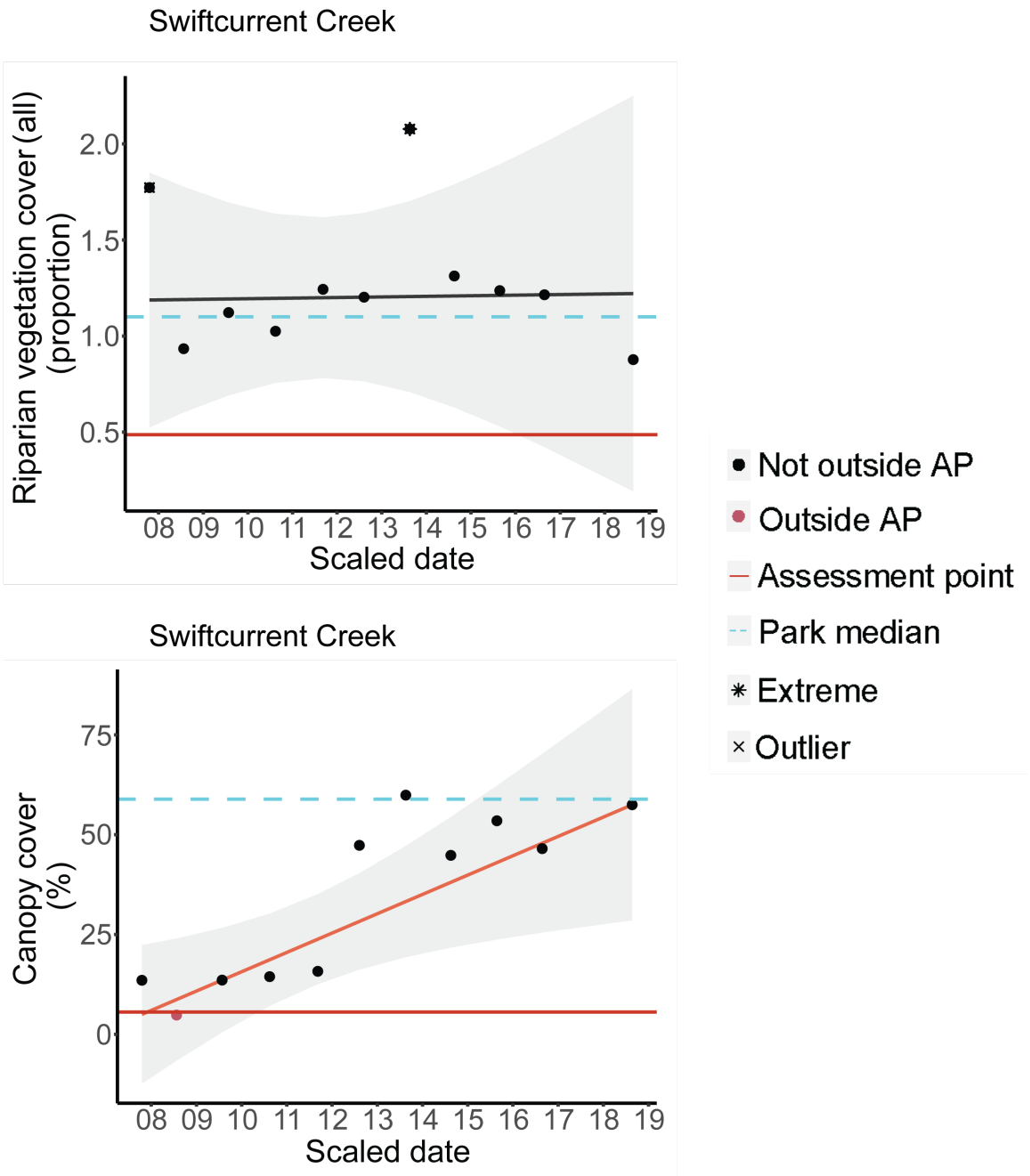


Figure 59. Examples of indicators of riparian vegetation cover in Glacier National Park from 2007 to 2019. (Top) riparian vegetation cover (all types) at Swiftcurrent Creek, and (bottom) total cover at Swiftcurrent Creek. As available, red horizontal lines show the non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (significant trend for canopy cover) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

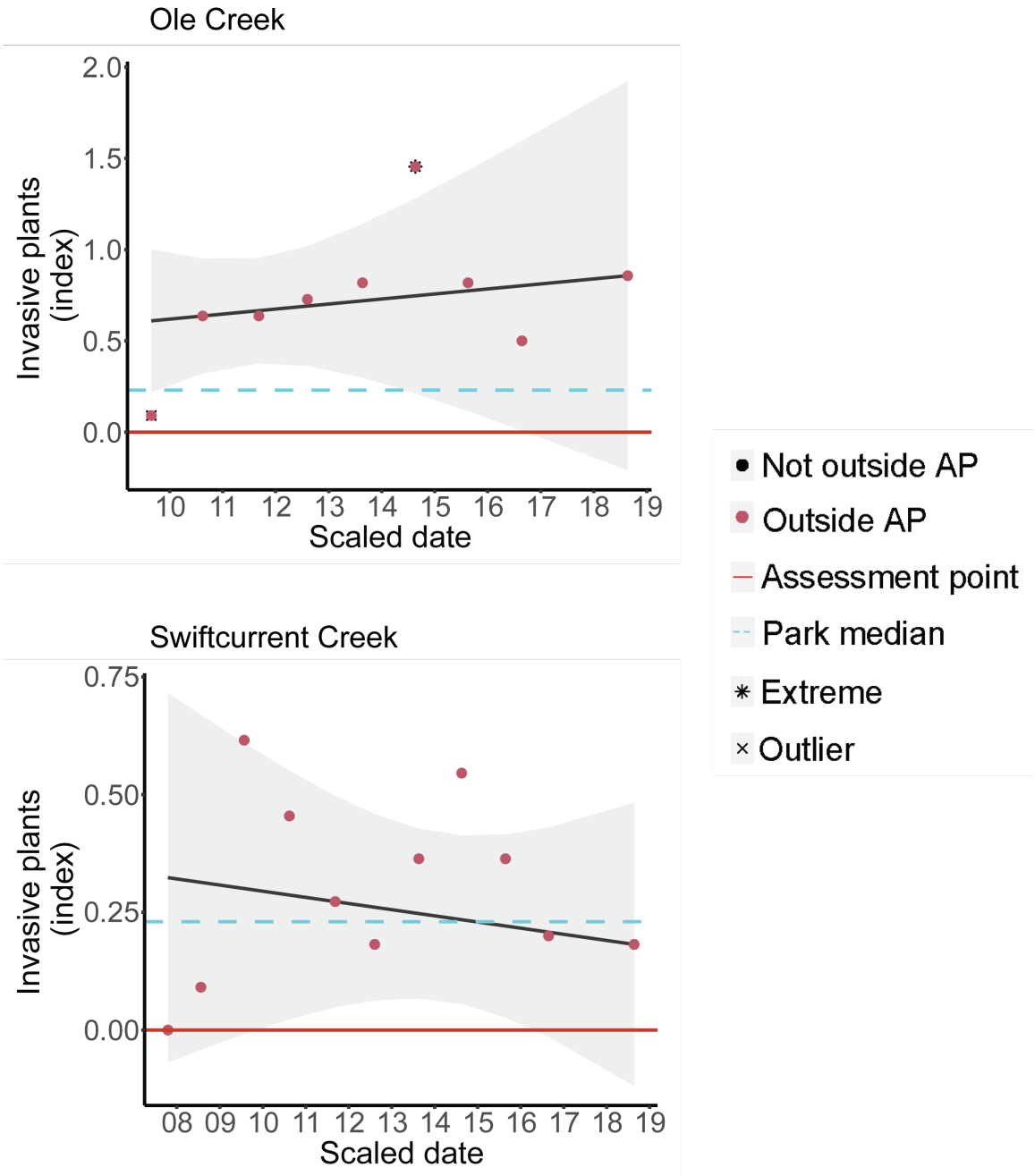


Figure 60. Example indicators of invasive/exotic plants in Glacier National Park from 2007 to 2019. (Top) Invasive Plants Index at Ole Creek, and (bottom) Invasive Plants Index at Swiftcurrent Creek. As available, red horizontal lines show the non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (no significant trends at these two sites) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

3.4.8.5 Condition

Very few events had lower riparian cover (total or woody) below the assessment point, suggesting that most sentinel stream sites usually had riparian cover above the best sites in the surrounding

ecoregions. Note that Reynolds Creek had a few riparian vegetation cover values below the reference range suggested by our assessment point. Reynolds Creek is in an alpine watershed where we expect little tree canopy and a reduced shrub layer; our assessment point is not ideal for such a context.

Total bank cover was below the assessment point in 40% of events on the North Fork Flathead—Border site. However, this is not surprising given the wide lateral bars that push bankfull height away from the riparian corridor where there is more vegetation cover. The few events at the other two large river sites were above the assessment point as these sites tended to have more riparian vegetation on the banks, with fewer in-channel lateral bars that separated the channel from the riparian zone.

Nearly all years at all sites except Reynolds Creek had invasive plants and were thus outside the park's conservative zero invasive/exotic plants assessment point. It is actually more surprising that there were a few years at some sites where this assessment point wasn't exceeded (meaning that we found no invasive/exotic plants). While the park's intention with setting this assessment point is reasonable, it is almost by definition going to lead to high percentages of values outside a reference condition.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead had riparian cover and invasive plant results like other sites.

3.4.8.6 Discussion and management applications

Riparian vegetation moderates stream temperatures through shading, increases bank stability, and provides coarse and fine particulate organic material (Naiman et al. 1988; Gregory et al. 1991). Organic inputs from riparian vegetation become food for stream organisms and provide structure that creates and maintains complex channel habitat. Results from most of our sentinel site sampling events suggest these functions are intact and we see no immediate issues beyond natural variability in the riparian systems at our sentinel sites.

Concerns about the repeatability of some of the visual estimates we use are legitimate given the challenging conditions under which estimates are often made, the semi quantitative nature of visual cover estimation, and the high variability in vegetation cover. Signal:noise estimates from the 2007–2009 survey for the indicators we use in this report are likely a useful approximation of the quality of sentinel sites' data. All vegetation and canopy cover estimates had acceptable or even quite high signal:noise, suggesting they are rigorous and represent the ecological response of interest.

One result of note is the consistently increasing total cover levels, which were significant and marginally significant at Swiftcurrent Creek and Reynolds Creek, respectively. Research has suggested that vegetation production across regional scales has been increasing, driven by elevated temperatures (i.e., Pouliot et al. 2009; Ichii et al. 2002). To explore this using our data, we estimated exploratory models that incorporated estimates of larger scale production at the sentinel sites from the normalized difference vegetation index and saw an improvement in the estimate of trend in canopy cover. While we lack sufficient data to make firm conclusions, it may be that we see this increased production in higher canopy cover along the banks of our sentinel sites. Whether this is a positive, negative, or inconsequential pattern will require additional data. Of note, we do not see this

increase in the visual riparian cover estimates at Swiftcurrent Creek. Total bank cover is measured with a densiometer and is a more quantitative measure. It also includes features such as hillslopes, large wood, or snags that can provide shade for a site.

Our overall impression is that the level of invasive/exotic plants at our sentinel sites during our period of record is not outside the norm for front country areas in the park (Lesica et al. 1993). Ole Creek might have more invasive vegetation than expected in some years given its more remote location. However, the site is on a trail that has a long history of stock use, some of which was prior to more strict controls on the feed used for stock and this may have allowed some invasive taxa to establish.

We have never recorded an invasive plant in the riparian corridor at the Reynolds Creek site. Alpine systems often have reduced levels of invasion given the short growing seasons and climatic conditions (i.e., Gallien et al. 2015). However, pilot work on long term wetland monitoring by the Rocky Mountain Network in Glacier National Park (McKernan et al. 2018) suggested climate driven changes in alpine hydrogeomorphic processes can trigger changes in vegetation composition by limiting establishment of disturbance-adapted species, such as *Salix* spp. Willows colonize bare mineral surfaces, and their reduction in the alpine might facilitate the establishment of upland and/or invasive species that also like bare exposed substrates. This motivates continued monitoring of invasive plants at this alpine sentinel site.

The processes driving invasion are complex and our work in this report just scratches the surface. Some research suggests that invasive plant species richness increases with native plant species richness at large scales (Allen et al. 2009). Thus, areas with exceptional diversity like Glacier—especially the riparian and wetland habitat in the park, with good growing conditions for native plants, such as sufficient light, water, and nutrients—may also provide good conditions for nonnative plant species (Stohlgren et al. 2001). Work by the network can complement ongoing efforts by the park's Invasive Plant Management program to address these questions into the future.

3.5 Bioassessment

3.5.1 Introduction

The assessment of biological data from stream monitoring has a rich history—even its own name "bioassessment." Bioassessment is the use of measures of a biological assemblage to assess and monitor natural systems (Barbour et al. 2000) and is the primary way we address objectives focused on estimating status in the condition (or ecology integrity) of Glacier National Park's sentinel stream sites. For clarity we use the term "bioassessment" for the process of using biology to understand ecological condition; "metrics" for individual biological variables; "indices" as synthetic variables often combined across metrics; and "bioindicators" as a more general term that is synonymous with both metrics and indices.

Two aquatic assemblages are central to the biology of Glacier's streams and rivers: 1) macroinvertebrates (Figure 61), including aquatic insects, worms, crustaceans, and other invertebrates, and 2) algae (Figure 61), or microscopic soft and hard bodied plants living in streams. Fish are of course also important, but SEI monitoring does not include this component.

Macroinvertebrates and algae are the base of the food web and thus support a broad range of other species. They are excellent indicators of ecological integrity because of their ability to integrate different processes and because they respond to physical, chemical, and biological disturbances at multiple temporal and spatial scales. Resh (2008) suggests that surveying macroinvertebrates is one of the most efficient ways to monitor streams over the long term, and macroinvertebrates are the most widely used group for biological monitoring of streams and rivers around the world (Bonada et al. 2006). For algae we focus on diatoms (single celled hard bodied alga with cell walls made of silica) and more specifically diatoms that live on substrates (also known as periphyton). Diatoms play important roles in the processing of CO_2, O_2, carbon, and silica in streams, among many other important ecological services. They are excellent biomonitors in a wide range of aquatic ecosystems (Stevenson and Smol 2003) because of their diversity, population sizes, short generation times, and high fidelity to specific ecological conditions.

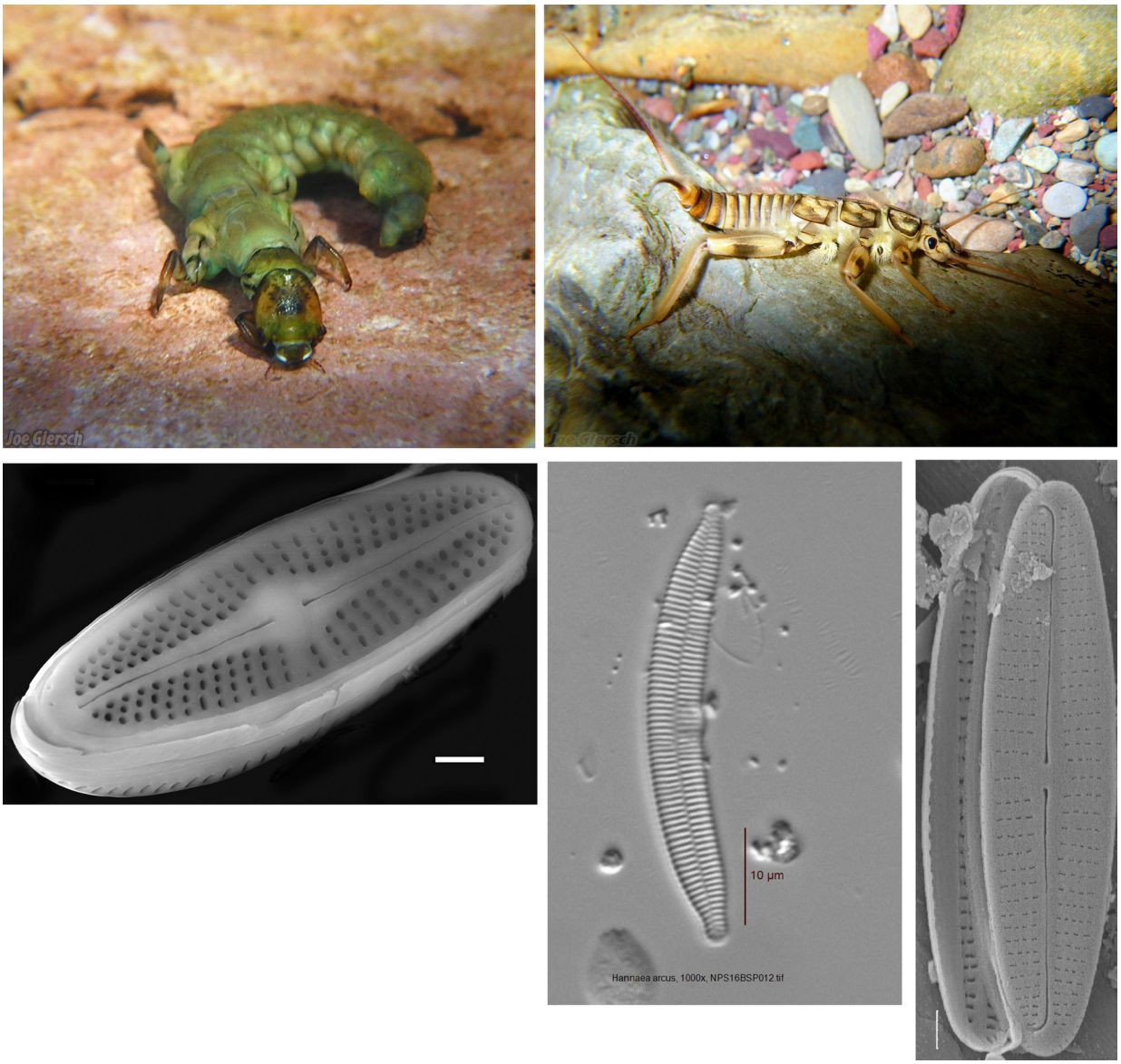


Figure 61. Example macroinvertebrate and diatom species in Glacier National Park. (Top left) *Rhyacophila* caddisfly typical in alpine streams like Reynolds Creek. (Top right) *Claassenia sabulosa* is a charismatic large predaceous stonefly, common in the park. Macroinvertebrate images courtesy J. Giersch, USGS. (Bottom left) *Achnanthidium minutissimum* is the most common diatom species in our park sample; image from Potapova, M. (2009b). (Bottom middle) *Hannaea arcus* from Swiftcurrent Creek, 2016, NPS image. (Bottom right) *Achnanthidium deflexum,* a sediment and nutrient increaser taxa; image from Potapova, M. (2009a).

Using both macroinvertebrates and diatoms is advantageous as each estimates different scales and types of ecological response to disturbance in the park's streams. Macroinvertebrates are among the most widely used organisms for bioassessment because they can be sampled relatively efficiently and effectively (Resh and Jackson 1993) (Figure 62); they are widespread in aquatic environments (Merritt et al. 2007); there are a large number of species that have a wide range of responses to environmental impacts (Resh et al. 1995); and since they are relatively sedentary, they can be used to

determine the spatial extent of impacts. In addition, since macroinvertebrates are relatively long-lived, community response can integrate the high temporal variability associated with traditional physical and chemical analyses (Rosenberg and Resh 1996). Diatoms have important functions in aquatic habitats as producers of organic matter and play a vital role in inorganic nutrient retention, transfer, and cycling (Stevenson and Smol 2003). As primary producers, diatoms are more sensitive to certain pollutants, like nutrients and herbicides, than other aquatic organisms. Different species are differentially sensitive to a variety of pollutants and have been found to be useful indicators of nutrient and clean-sediment impacts. Measures of the structure of algal associations, such as species diversity and dominance, can be sensitive and useful indicators of water-quality impacts and ecological disturbance. Diatoms are useful indicators of environmental condition because they respond rapidly and are sensitive to several anthropogenic disturbances, including habitat destruction and contamination by nutrients, metals, herbicides, hydrocarbons, and acidification (e.g., Hill et al. 2003). In streams where flow and substratum characteristics create efficient interactions between water and the benthic diatom assemblage (periphyton), diatoms typically reflect recent water chemistry (Lowe and Pan 1996). Diatom assemblage composition is also strongly influenced by land–water interactions, and by stream size and the level of human disturbance.



Figure 62. Volunteer monitoring crew from the US Forest Service collecting macroinvertebrates on the North Fork of the Flathead River, Glacier Rim, 2021. (NPS / B. SCHWEIGER)

We also include results from our monitoring for Aquatic Invasive Species (AIS). These taxa can have profound effects on streams and rivers and are thus of concern for Glacier National Park resource management. They may also be synthetic indicators of non-reference conditions in a stream site. While the sample design of SEI monitoring is not explicitly geared towards collecting data on AIS (i.e., we do not target sites where invasion is more likely), samples across time at sentinel sites can provide useful data to help the park understand potential occurrences of important AIS. Given constraints in our protocol, SEI data should not replace more focused efforts in Glacier for monitoring and management of these species. Glacier National Park resource management and the Rocky Mountain Network decided to focus on four species: rock snot (*Didymosphenia geminata*), zebra mussels (*Dreissena polymorpha*), quagga mussels (*Dreissena rostriformis bugensis*), and New Zealand mud snails (*Potamopyrgus antipodarum*). There are other AIS of major importance, especially nonnative fish (C. Downs, NPS biologist, pers. comm. 10/2022), but this subset fit our methods well.

3.5.2 Summary condition

Summary condition statements condense bioassessment results into concise overviews of key aspects of trends, assessment, and possible management applications or concerns (Table 25).

Table 25. Summary condition table for biological indicators at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Bioassessment (overall)	5 indicators	The macroinvertebrate and diatom communities of Glacier National Park are some of the most diverse and intact examples known in the world. We utilize these communities as bioassays, and these models and interpretations are the primary ways in which we estimate status in the condition (or ecology integrity) of Glacier's sentinel stream sites. In this report we focus on stressor specific macroinvertebrate and diatom indices. It is useful to directly link macroinvertebrate response to important specific stressors in Glacier. This can be diagnostic and thus able to support more specific resource management needs. Future work will present an updated synthetic multimetric index for macroinvertebrates and diatoms. Overall, our bioindicators of specific stressors in the park suggested some caution, but in general, this is isolated, and we retain a reference assessment for the category. However, there are strong responses in our bioindicator of water temperature, once again showing the importance of the park's warming waters. There is also a fair degree of biological response to nutrients, yet when combined into the state's integrated nutrient protocol, we do not see nutrient impairment at any sentinel site over our period of record. Some biological results, when contrasted with chemistry or habitat indicators of a similar attribute, relaxed the assessment of a non-reference condition based on the chemical or habitat response. This is most notable for fine sediments, where we do not see as strong of a biological response as we do in actual habitat-based sediment data. Our confidence in our overall bioassessment is medium given some unclear trends.	
Macroinvertebrates	Fine Sediment Biotic Index	We estimate a biological response to sediments in macroinvertebrates at our sentinel sties with the Fine Sediment Biotic Index (FSBI). There were no significant or marginally significant trends in FSBI at any sentinel site. The directions of slopes varied with site, but across all sites, there was no change over time in the biological response to fine sediments over our period of record. There are no management, regulatory, or known ecological assessment points for the FSBI. We do not have estimates of this index in the usual ancillary ecoregional reference site data used throughout this report and instead use data reported in the literature and a relaxed assessment point threshold percentile. FSBI at Swiftcurrent Creek was below the assessment point of 15.4% of our samples. At the North Fork Flathead—Border site, macroinvertebrate assemblage composition suggested there were excess fines at the site 9.1% of the time. Potentially problematic values at Swiftcurrent Creek may have been associated with a beaver dam and/or the higher visitor use of the site; additional work is needed on this. The biological response seen in FSBI at Swiftcurrent Creek, and to a lesser degree at the North Fork Flathead—Border, mirrors patterns in sediment data. While no trends were significant, several results are suggestive. However, with only a few FSBI data points in a non-reference condition, we have relaxed our interpretation of a possible sediment issue at Swiftcurrent Creek based on habitat data. We feel additional data and monitoring are warranted. Should fine sediments become more prevalent at sentinel sites or a more severe biological response seen, the park may explore restoration or remediation at this site. We also see an interpretable relationship between FSBI and the Human Disturbance Index (HDI), with lower FSBI scores (more sensitive tolerant taxa) in sites with higher general human disturbance. While this needs more careful treatment, it is suggestive of a human disturbance signal driving a sediment response in macroinvertebrates. Our confidence in FSBI is medium given some unclear trends.	

Table 25 (continued). Summary condition table for biological indicators at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

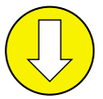
Vital sign	Indicator	Summary condition	Status, trend, and confidence
Macro-invertebrates (continued)	Temperature Preference Index	We estimate a biological response to sediments at our sentinel sties with the Temperature Preference Index (TPI). Water temperature is difficult to measure and an index like the TPI provides a useful bioassay of temperature dynamics in a stream. In a sense the TPI treats the macroinvertebrate assemblage as a natural temperature logger that integrates highly variable water temperatures experienced across the life cycles of the taxa in a stream. The TPI increased at all sites except Ole Creek over our period of record. The North Fork Flathead—Border site had a strongly significant increasing trend—one of the strongest in our entire dataset—indicating a "warmer" macroinvertebrate assemblage developed over time at this site. There are no management, regulatory, or known ecological assessment points for TPI. We estimate an ecoregional assessment point from reference sites that provides general guidance for Glacier National Park bioassessment of water temperature under an expectation of a reference state. Interestingly, this assessment point (15 °C) is very similar to the ecological thresholds developed for water temperature proper. TPI values approached the assessment point at most sites but did not exceed the assessment point, though all were headed in that direction. Models of continuous logger-based temperature also show increasing water temperature, often related to stream flow, soil moisture, and evapotranspiration (see the General Additive Model results in Appendix C). Direct temperature results had a clear non-reference condition; however, at least based on the TPI index at our sentinel sites, the biological response may not be as dramatic, or it has a lagged relationship with water temperatures. We thus rate this as yellow, or caution warranted. Exploratory work indicates a rich array of connections between TPI and site-level water chemistry, habitat, and other biological measures. For example, TPI increases with specific conductance, suggesting more high temperature tolerant macroinvertebrates persist in less dilute waters that are often in lower elevation larger rivers (that are relatively warmer). We also see more high temperature tolerant macroinvertebrates in systems with higher pool density. The slack water of pools is typically warmer. These will need further exploration, but they support both the structural, wide ranging role of water temperature in stream ecology, and suggest future models that can capture these relationships in a predictive context. Our confidence in our TPI data, and in our assessment of patterns in these data, is medium.	

Table 25 (continued). Summary condition table for biological indicators at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Macro-invertebrates (continued)	Hilsenhoff Biotic Index (nutrients)	Excess nutrient levels are another stressor to Glacier National Park sentinel stream sites featured throughout this report. Nutrients are a primary focus of Montana DEQ regulatory work and there is a long history of research on the trophic status of aquatic systems. A bioindicator approach to nutrients facilitates integration across many of the complexities inherent in water chemistry based nutrient monitoring. We use a classic Hilsenhoff Biotic Index (HBI) as our bioassay of nutrients. HBI at the North Fork Flathead—Border site decreased significantly, indicating that a macroinvertebrate assemblage with fewer taxa tolerant of elevated nutrients and/or more taxa sensitive to nutrients developed over time—this is somewhat counter to most other trends in nutrients where we see degrading condition. Additional work is needed to resolve this. Other sites had mixed responses in HBI, none of which were significant. For the HBI assessment point, we use a Montana DEQ regulatory criteria developed using ecoregional data and change-point analysis. The state uses this index and its criterion as a component of an integrated nutrient assessment (that we also apply to our sentinel site data). This is likely the best example we have of bioassessment playing a key role in regulatory (or otherwise) monitoring. HBI was often above its assessment point, especially at the Swiftcurrent Creek and North Fork Flathead—Border sites. However, HBI also dropped across our period of record at these two sites, with individual values above the assessment point and in non-reference condition near the end of data, even as the overall trend decreased. We also see higher levels of total phosphorous at the North Fork Flathead—Border site in our water chemistry data—but not as high as with HBI, suggesting some amplification of a nutrient effect in the biological response. Importantly, using the Montana DEQ integrated nutrient assessment protocol, this combination still does not suggest a nutrient impairment and there is no known human source of nutrients in or near our sentinel sites. However, overall, for a wilderness park, this may nevertheless warrant some caution. Our confidence in our HBI data, and in our assessment of patterns in these data, is medium.	

Table 25 (continued). Summary condition table for biological indicators at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Diatoms	Nutrient increasers	We estimate a biological response to nutrients in diatoms using the Montana DEQ nutrients increasers index. This index is based on the percent relative abundance of diatom taxa that exist in detectable amounts in stream types defined by high levels of nutrients. These diatoms are considered ecological specialists that are tolerant of excess nutrients. There were no significant or marginally significant trends in nutrient increasers at any sentinel site; however, and interestingly, the directions of slopes were negative across all sites. This suggests a consistent reduced biological response in diatoms to nutrients. Montana DEQ specifies a criterion for nutrient increasers corresponding to the 50% probability of nutrient impairment and uses the index as part of an integrated methodology focused on nutrient impairment. We use the nutrient increaser criterion as an assessment point, but regulatory authority rests with Montana DEQ and its interaction with Glacier National Park resource management. Our nutrient increaser results do not suggest impairment, with perhaps an exception at the North Fork Flathead—Border site. We see an interesting pattern between phosphorous and nitrogen concentrations and nutrient increasers. Phosphorous consistently has stronger and positive relationships with nutrient increasers, while nitrogen parameters are all negative. This suggests that nutrient sensitive diatoms are likely limited by phosphorous, and when it has a higher concentration, these species respond with higher relative abundances. This has become an important part of how regional scale patterns in nutrients are understood and will likely be central to our continued work on nutrients in Glacier. Our confidence in our diatom nutrient increaser data, and in our assessment of patterns in these data, is medium.	

Table 25 (continued). Summary condition table for biological indicators at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Diatoms (continued)	Sediment increasers	We estimate a biological response to sediments in diatoms using the Montana DEQ sediment increasers index. As with nutrients, this is based on the percent relative abundance of diatom taxa that exist in detectable amounts in stream types defined by high levels of fine sediments. There was one significant trend in sediment increasers at Swiftcurrent Creek, with the probability of sediment impairment decreasing over time (i.e., the biological response to fine sediments at the site decreased, suggesting fewer fines). The directions of slopes at other sites (except Reynolds Creek) were also negative, indicating sediment increasers were (nonsignificantly) decreasing. We thus rate this as an increasing trend in condition. Montana DEQ specifies a criterion for sediment increasers corresponding to a 50% probability of fine sediment impairment. Currently the index is not part of a larger integrated methodology focused on sediment impairment. Montana DEQ is working on this, and we will adopt it as applicable in the future. We use the MT DEQ criterion as an assessment point. Several sediment increaser results were outside the reference range suggested by this assessment point. However, these were single events within each site's period of record. In nearly all cases, the prevalence of increaser diatom taxa was moving away from the Montana DEQ criteria defined probability of impairment at each site. The responses at the North Fork Flathead—Border and Reynolds Creek sites were nearly above the assessment point in a second event, suggesting that, at these sites, diatoms were slightly more reflective of increased sediments. Decreasing trends in sediment increasers at most sentinel sites was counter to increasing or flat fine sediment results in our habitat data and in the macroinvertebrate sediment response. We interpreted the actual fine sediment data as likely due to the beaver dam, with some possibility that visitor use, or related support facilities, also introduced more fines to the site. Increased chloride and calcium ion concentration also speak to a possible human disturbance source at Swiftcurrent Creek. The lack of a biological response in at least diatoms, and decreasing trends in both assemblages, may suggest either no strong biological effect of the higher fines or a lag in the relationship. We feel additional data and monitoring is warranted; should fine sediments become more prevalent in the site or a more severe biological response seen, the park may explore restoration or remediation at this site. Our confidence in our diatom sediment increaser data, and in our assessment of patterns in these data, is medium.	
Aquatic invasive species	Didymo	Aquatic Invasive Species (AIS) can have a strong effect on or reflect stream ecological integrity. They are a focus of park resource management. The invasive but native diatom didymo (also known a rock snot; *Didymosphenia geminata*) was found in a large portion of the park's streams in our 2007–2009 baseline, often in a bloom state. However, sentinel sites were generally not part of the parkwide high levels in 2007–2009. This low presence across all sentinel sites continued through 2019, with no significant trends at any site. However, it may be premature to say that didymo no longer is an issue at the park scale based on our sentinel sites alone. Additional data, especially a resample of the full array of survey and gradient sites, is needed to know what didymo has done over the last decade in the park. Our confidence in our didymo data, and in our assessment of patterns in these data, is medium. Recent research has better documented the negative impacts of rock snot on stream condition. We still feel that given these potential consequences, this is an issue of importance to the park.	

Table 25 (continued). Summary condition table for biological indicators at Glacier National Park in 2007–2019. Key to the status, trend, and confidence symbol: red color = non-reference condition/warrants significant concern, yellow color = intermediate condition/warrants moderate concern, green color = reference/good condition; upward arrow = condition is improving, sideways arrows = condition is unchanging, downward arrow = condition is deteriorating; heavy border = high confidence, medium-weight border = medium confidence, dashed border = low confidence. The direction of arrows refers to the change in condition, not of an indicator's numeric value.

Vital sign	Indicator	Summary condition	Status, trend, and confidence
Aquatic invasive species (continued)	Other non-fish aquatic invasive species	We did not detect zebra mussel (*Dreissena polymorpha*), quagga mussel (*Dreissena rostriformis bugensis*), or New Zealand mudsnails (*Potamopyrgus antipodarum*) during our parkwide survey in 2007–2009 or at our sentinel sites between 2007 and 2019. Monitoring should continue to watch for these and other invasive aquatic species. Our confidence in our AIS data, and in our assessment of patterns in these data, is medium.	

3.5.3 Macroinvertebrate and algae sentinel site diversity

From 2007 to 2019 we identified 686 diatom and 435 macroinvertebrate taxa across all sites (including survey and gradient sites; Table 26; Appendix D). At our six sentinel sites, 544 of these diatom taxa and 356 macroinvertebrate taxa were collected. This represents 79% and 81% of our total diatom and macroinvertebrate taxa counts, respectively, indicating a substantial portion of the diversity of these two assemblages occurs at sentinel sites in the park. An additional 73 soft bodied algae taxa have been identified across all Glacier National Park sample events, with 55 of these recorded in sentinel sites from 2007 to 2019.

Table 26. Summary of diversity indices for macroinvertebrates and diatoms monitored at sentinel sites in Glacier National Park, 2007–2019.

Indicator	Total taxa richness	Mean Shannon diversity	Mean taxa richness
Diatoms	544	3.56	45.9
Macroinvertebrates	356	4.01	52.9

Mean average taxa richness for all sentinel site events was 49.7 and 53.2 for diatoms and macroinvertebrates, respectively. The highest sentinel site diatom and macroinvertebrate taxa richness was 93 and 80, collected at Swiftcurrent Creek in 2011 and 2018, respectively, while the lowest was 13 and 22 collected from Reynolds Creek in 2015 for diatoms and at the North Fork Flathead—Border site in 2008. Mean Shannon diversity (taxa counts weighted by their relative abundance) was 3.56 and 4.01 for diatoms and macroinvertebrates in sentinel sites, respectively. Shannon diversity is generally between 1.5 and 3.5 in most ecological studies, and the index is rarely greater than 4 (an assemblage with only one species would have a value of 0). The Shannon index increases as both the richness and the evenness (or numbers of individuals within a taxa) of the assemblage increase. Glacier National Park diversity measures are typical of taxa rich assemblages, with a pronounced inverse J curve in species accumulation, indicating that many sites have a few species with high relative abundances, and a longer list of taxa with low relative frequencies of occurrence across sites.

Note that richness may be confounded with sample effort to some degree, but most sample events were comparable, and all lab methods were consistent (i.e., a common target count of individuals was used). Shannon diversity is also corrected for richness. We feel these simple metrics are thus generally valid; however, more detailed work, such as rarefaction, is needed to fully resolve this.

Diatoms were almost always identified to species, while the identification level for macroinvertebrates was a mix of species and genera and higher level operational taxonomic units (OTU; Jessup et al. 2006; Montana DEQ 2012a). In 2007 through part of 2009, we used a slightly coarser macroinvertebrate OTU. Analyses and models in Schweiger et al. (2018) were based on this nomenclature. In this report, we have cross walked all macroinvertebrate identifications to a new system that includes finer level OTU names (and various other taxonomic updates). In all cases, non-species level identifications were included in analyses only if we had confidence that an

identification for a taxon was consistent across all samples. Appendix D gives complete taxa lists for all assemblages.

In qualitative comparisons to Environmental Monitoring and Assessment Program (EMAP) data from the Northern and Canadian Rockies (Stoddard et al. 2005), the diversity of Glacier's macroinvertebrate assemblage was high. Across all sentinel sites, mean diversity in Glacier was 3.56; for ecoregion reference sites, the site level average was around 2.8. The World Wildlife Fund rates the biological distinctiveness of the Canadian Rockies Ecoregion as "Continentally Outstanding" (Abell et al. 2000). Much of the Pacific Northwest, within which the Canadian Rockies lies, was heavily glaciated during the Pleistocene and as a result has few known endemic aquatic species. However, the Canadian Rockies Ecoregion served as a refugium for freshwater species during the last glaciation, and as a result, the region supports a number of endemic fish and invertebrates (Abell et al. 2000). Glacier National Park, at the headwaters of three continental drainages, protects and provides clean water and high quality aquatic habitats for these species. Stanford et al. (1994) describes the macroinvertebrate assemblages of the park as very species rich, owing in part to the pristine, oligotrophic nature of the waters and the wide variety of aquatic habitats along an altitudinal gradient. Moreover, because the park exists midway in the north–south continuum of the Rocky Mountains, and three major river systems of the North America continent share adjacent headwaters, many species are on the fringe of their continental distribution and the area serves as a "melting pot for biodiversity." For example, 67% (101 species) of Plecoptera (stoneflies) reported for the Rocky Mountains from Alaska to New Mexico have been collected in the Flathead River catchment.

The diatom assemblage in Glacier National Park is thought to be one of the more diverse assemblages known (Spaulding et al. 2010). If lakes, wetlands, seeps, and other standing water habitats are included, along with flowing waters, the total number of recognized diatom species in the park is likely well over 1,000 (L. Bahls, algae expert, retired, pers. comm., 10/2012). Historically, the most completely studied country for diatoms is Germany, where 1632 species have been reported. Therefore, Glacier, which is, of course, much smaller (Glacier is 1.5% the size of Germany) and lacks estuarine and brackish water habitats, has over two-thirds the number of diatom species of this large and diverse country. Importantly there is a sampling intensity effect here, with other remote and previously unexplored regions of the world having comparable diatom diversity when sampled as thoroughly as Glacier (e.g., Finland, Madagascar, Sardinia, the Andes, Arctic Canada).

Table 27 summarizes macroinvertebrate metrics analyzed at sentinel stream monitoring sites in the park.

Table 27. Select macroinvertebrate indicators at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside (or in exceedance if a regulatory criteria assessment point (AP) type) is the percentage of results above (and/or below) the given assessment point(s). Any Percent outside value above or equal to 25% is in bold. AP types include regulatory criteria (CRT) and ecoregional (ERG). The fine sediment biotic index increases as condition in a stream increases, and values below or equal to the AP are in a non-reference condition. The Hilsenhoff Biotic Index and Temperature Preference Index increase as condition in a stream deteriorates, and values above or equal to the assessment points are in a non-reference condition. For the temperature preference index ecoregional assessment points, we use the 5th percentile value from 48 ecoregional reference sites. Following Stoddard et al. (2008), Fine Sediment Biotic Index ecoregional assessment points use the 25th percentile value from 428 sites in Northern Mountain ecoregions (Cascades, N. Rockies, ID Batholith; Relyea et al. (2012)). For the Hilsenhoff Biotic Index, we use a criteria from Montana DEQ (Suplee and Sada 2016).

Metric	Site	Trend in partial residual	Trend (p)	Percent outside / in exceedance of AP	AP	AP Type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Fine Sediment Biotic Index (2012)	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	105	ERG	8/25/2007	8/20/2019	4	120.0	140.0	135.000	27.839	105.0	160
	North Fork Flathead—Border	2.4974	0.621	9.1	105	ERG	8/20/2007	8/22/2019	11	120.0	160.0	148.636	32.333	100	180
	Swiftcurrent Creek	−2.4949	0.686	15.4	105	ERG	10/18/2007	8/22/2019	13	120.0	130.0	136.923	48.885	65	230
	Reynolds Creek	0.4962	0.786	0.0	105	ERG	9/11/2008	8/20/2019	12	120.0	127.5	129.167	13.286	110	160
	Ole Creek	4.3795	0.205	0.0	105	ERG	8/27/2009	8/21/2019	11	120.0	165.0	167.273	22.623	135	210
	Middle Fork Flathead	– [a]	– [a]	0.0	105	ERG	8/28/2018	8/19/2019	2	120.0	130.0	130.000	35.355	105	155
Temperature Preference Index	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	15.22	ERG	8/25/2007	8/20/2019	4	11.8	14.379	14.435	0.358	14.108	14.817

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend is significant ($p \leq 0.10$).

[c] Percent outside value is ≥ 25%.

Table 27 (continued). Select macroinvertebrate indicators at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside (or in exceedance if a regulatory criteria assessment point (AP) type) is the percentage of results above (and/or below) the given assessment point(s). Any Percent outside value above or equal to 25% is in bold. AP types include regulatory criteria (CRT) and ecoregional (ERG). The fine sediment biotic index increases as condition in a stream increases, and values below or equal to the AP are in a non-reference condition. The Hilsenhoff Biotic Index and Temperature Preference Index increase as condition in a stream deteriorates, and values above or equal to the assessment points are in a non-reference condition. For the temperature preference index ecoregional assessment points, we use the 5th percentile value from 48 ecoregional reference sites. Following Stoddard et al. (2008), Fine Sediment Biotic Index ecoregional assessment points use the 25th percentile value from 428 sites in Northern Mountain ecoregions (Cascades, N. Rockies, ID Batholith; Relyea et al. (2012)). For the Hilsenhoff Biotic Index, we use a criteria from Montana DEQ (Suplee and Sada 2016).

Metric	Site	Trend in partial residual	Trend (p)	Percent outside / in exceedance of AP	AP	AP Type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Temperature Preference Index (continued)	North Fork Flathead—Border	0.1271	**0.050** [b]	0.0	15.22	ERG	8/20/2007	8/22/2019	11	11.8	13.488	13.611	0.682	12.133	14.441
	Swiftcurrent Creek	0.0907	0.161	0.0	15.22	ERG	10/18/2007	8/22/2019	13	11.8	12.977	13.108	0.683	12.211	14.676
	Reynolds Creek	0.0744	0.289	0.0	15.22	ERG	9/11/2008	8/20/2019	12	11.8	10.830	11.021	0.560	10.491	12.558
	Ole Creek	−0.0064	0.754	0.0	15.22	ERG	8/27/2009	8/21/2019	11	11.8	11.221	11.224	0.128	10.98	11.448
	Middle Fork Flathead	– [a]	– [a]	**50.0** [c]	15.22	ERG	8/28/2018	8/19/2019	2	11.8	15.075	15.075	0.501	14.72	15.429
Hilsenhoff Biotic Index (nutrients)	North Fork Flathead—Glacier Rim	– [a]	– [a]	**50.0** [c]	4	CRT	8/25/2007	8/20/2019	4	3.4	4.750	4.250	0.900	3.21	4.789
	North Fork Flathead—Border	−0.1386	**0.056** [b]	**45.5** [c]	4	CRT	8/20/2007	8/22/2019	11	3.4	3.925	4.019	0.607	3.209	5.041
	Swiftcurrent Creek	0.0685	0.526	23.1	4	CRT	10/18/2007	8/22/2019	13	3.4	3.161	3.287	0.843	2.252	4.655

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend is significant ($p \leq 0.10$).

[c] Percent outside value is ≥ 25%.

Table 27 (continued). Select macroinvertebrate indicators at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10). Percent outside (or in exceedance if a regulatory criteria assessment point (AP) type) is the percentage of results above (and/or below) the given assessment point(s). Any Percent outside value above or equal to 25% is in bold. AP types include regulatory criteria (CRT) and ecoregional (ERG). The fine sediment biotic index increases as condition in a stream increases, and values below or equal to the AP are in a non-reference condition. The Hilsenhoff Biotic Index and Temperature Preference Index increase as condition in a stream deteriorates, and values above or equal to the assessment points are in a non-reference condition. For the temperature preference index ecoregional assessment points, we use the 5th percentile value from 48 ecoregional reference sites. Following Stoddard et al. (2008), Fine Sediment Biotic Index ecoregional assessment points use the 25th percentile value from 428 sites in Northern Mountain ecoregions (Cascades, N. Rockies, ID Batholith; Relyea et al. (2012)). For the Hilsenhoff Biotic Index, we use a criteria from Montana DEQ (Suplee and Sada 2016).

Metric	Site	Trend in partial residual	Trend (p)	Percent outside / in exceedance of AP	AP	AP Type	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Hilsenhoff Biotic Index (nutrients) (continued)	Reynolds Creek	0.0912	0.206	0.0	4	CRT	9/11/2008	8/20/2019	12	3.4	2.255	2.407	0.617	1.525	3.53
	Ole Creek	0.0073	0.873	0.0	4	CRT	8/27/2009	8/21/2019	11	3.4	1.532	1.538	0.350	0.872	2.032
	Middle Fork Flathead	– [a]	– [a]	**50.0** [c]	4	CRT	8/28/2018	8/19/2019	2	3.4	4.153	4.153	1.283	3.245	5.06

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend is significant ($p \leq 0.10$).

[c] Percent outside value is ≥ 25%.

3.5.4 Macroinvertebrates

The following sections present results and interpretation of select macroinvertebrates bioassessment metrics at sentinel sites sampled in 2007–2019. Additional results, specifically an update to the synthetic Glacier Multimetric Index for macroinvertebrates, are currently under production and will be presented in a future publication. Here we focus on patterns over time in three stressor-specific metrics.

Bioassessment via indices linked to specific stressors is a useful complement to more general synthetic tools such as the multimetric index (MMI) presented in Schweiger et al. (2018). Stressor specific bioindicators reveal if a given stressor "matters" to a component of the biology in a sentinel stream site. In most cases, additional work is needed to know that these linkages are causal or structural, but relatively simple correlations, if supported by sound reasoning, are often effective in making useful interpretations (Grace and Irvine 2020). Stressors like fine sediment or excess nutrients rarely occur in isolation and have complex interactions. A stressor specific bioindicator might still show a biological response even in the face of this complexity (Friberg 2010); however, a general or synthetic index might also deal with these interactions by, in a sense, averaging across them—hence our use of both approaches. Bioassessment might be advanced by the development of diagnostic methods to identify the stressors causing ecological impact rather than to only infer impact intensity and assign quality ratings to assessment sites (Chessman 2021). Montana DEQ's use of the HBI and diatom increaser indices is an excellent example of a real world application of bioassessment to complete the story on possible nutrient impairment. These three bioindicators were well suited to our Glacier National Park data, with tolerance values populated for 27%, 51%, and 84% of the taxa for FSBI, TPI and HBI, respectively. The taxa within the less well attributed FSBI (and to a lesser degree, TPI) metric were abundant, improving the representativeness of the FSBI for the sites where these common taxa were recorded.

3.5.4.1 Background and relevancy

Sediment: Fine Sediment Biotic Index

The Fine Sediment Biotic Index (FSBI; Relyea et al. 2000, 2012) is a tolerance-based metric that describes the response within a benthic assemblage to fine sediment levels. The original model was developed for specific ecoregions, including the two in which Glacier National Park occurs. Taxa from 428 streams were scored based on their occurrence along a gradient of fine sediment levels; taxa found with lower levels of fine sediment have higher scores. The FSBI is calculated as the sum of the scores of all taxa in a sample. It ranges from 0 to ~350, with higher scores corresponding to sites with more taxa sensitive to higher levels of fine sediments (i.e., higher scores indicate sites with lower fine sediment levels, or a more reference like condition).

Temperature: Temperature Preference Index

The Temperature Preference Index (TPI; Brandt 2001) was derived from estimates of temperature optima for a suite of 137 macroinvertebrate taxa sampled at 2055 sites in the Northern/Canadian Rockies ecoregion (within which Glacier is embedded). The index is estimated as the mean temperature for taxa that exhibited strong responses to empirical temperature gradients across sites.

TPI ranges (theoretically) from ~0 to ~25. The standard deviation of the observed temperature range is also available in Brandt (2001) and can be used as a tolerance to temperature changes.

Nutrients: Hilsenhoff Biotic Index

We use the Hilsenhoff Biotic Index (HBI; Hilsenhoff 1977, 1987) as an estimate for tolerance to nutrients, or more generally, organic enrichment. The HBI index has been in widespread use for several decades and is itself often integrated into MMI models. The HBI uses scores for taxa developed by many sources over the years that estimate a tolerance to organic nutrient pollution. Taxa are assigned a tolerance number from 0 to 10 pertaining to the taxa's sensitivity to organic pollutants. A score of 0 is the most sensitive, while a score of 10 is the most tolerant (i.e., sites with high HBI scores have higher organic pollution). The final HBI is calculated as the weighted sum of taxa in a sample adjusted by the total number of individuals counted (with the weight being the HBI tolerance score).

3.5.4.2 Assessment points

There are no management, regulatory, or known ecological assessment points for the FSBI or TPI indices, whereas a relevant regulatory criterion is available for HBI. We develop assessment points for these metrics as follows.

For the TPI, we estimate an ecoregional assessment point from 48 ecoregional reference sites (Table 27). Values above or equal to this assessment point are in a non-reference condition. Interestingly, the TPI ecoregional assessment point, at 15.2 °C, is nearly identical to the suite of bull trout and macroinvertebrate ecological thresholds used for continuous temperature data as presented above.

For the FSBI, we do not have estimates of this index in our ancillary data sites (see map of ecoregional assessment points in Appendix A) used throughout this report and instead use data reported in Relyea et al. (2012). These data are not classified or restricted to reference sites as required by our assessment methodology. Stoddard et al. (2008) recommends in these situations to relax the assessment point threshold to the 25th percentile (from the 5th). For the FSBI, values below or equal to the assessment point of 105 are in a non-reference condition.

For the HBI, Montana DEQ (2012a) presents an analysis of Montana data examining the relationships between stream nutrient concentrations and benthic macroinvertebrate metrics (see also Li et al. 2010; Suplee and Sada de Suplee 2011). Among the metrics examined, HBI showed a sufficiently patterned response to stream nutrient gradients in Montana's mountainous regions that Montana DEQ uses it as a secondary response variable to help assess nutrient impacts (Suplee and Sada de Suplee 2016; Suplee and Watson 2013). Montana DEQ applied change-point analysis (Qian et al. 2003) to statistically derive a shift or threshold in HBI and nutrient concentrations at a threshold value of 4.0. We use this criterion as an assessment point in our work, but again any regulatory authority related to it rests with Montana DEQ and its interaction with Glacier National Park resource management. Values above or equal to this assessment point are in a non-reference condition.

3.5.4.3 Parkwide 2007–2009 baseline results

Schweiger et al. (2018) found that 11.7% of the streams in the park in 2007–2009 had a non-reference FSBI. This was roughly equivalent to the amount of stream length with high levels of measured fine sediments. This suggests an expected or roughly equal biological response to smaller sediments at sentinel sites. Most sites with higher sediment levels were in or near the front country, suggesting a more likely potential human source. Sources may have included roads or other visitor services infrastructure. Sentinel sites (largely in the front country) in 2007–2009 generally had reference level FSBI values. The biological response to water temperature in 2007–2009 at the park scale was mild, with only 3% of the park streams above the assessment point in 2007–2009. There was a strong response in TPI to both instantaneous measured water temperature and modeled catchment maximum water temperatures (Jones et al. 2014, 2017) suggesting the TPI index was a useful bioindicator of actual stream water temperatures in the park. The biological response in the macroinvertebrate assemblage to nutrients via the HBI was more pronounced in the 2007–2009 baseline survey. Around 30% of the park's flowing water was in a non-reference state and several sentinel sites had elevated HBI values in the baseline period.

3.5.4.4 Trend

There were no significant or marginally significant trends in FSBI at any sentinel site (Table 27, Figure 63). The directions of slopes varied with site, but across all sites, there was no change over time in the biological response to fine sediments over our period of record. However, there were significant trends in TPI and HBI (Table 27). TPI at site North Fork Flathead—Border increased significantly (Figure 64) indicating a "warmer" macroinvertebrate assemblage developed over time. This increase may reflect elevated water temperatures at this sites as presented in this report. HBI at this site decreased significantly (Figure 65), indicating a macroinvertebrate assemblage with fewer taxa tolerant of elevated nutrients and/or more taxa sensitive to nutrients developed over time. Other sites had mixed responses in TPI and HBI, none of which were significant.

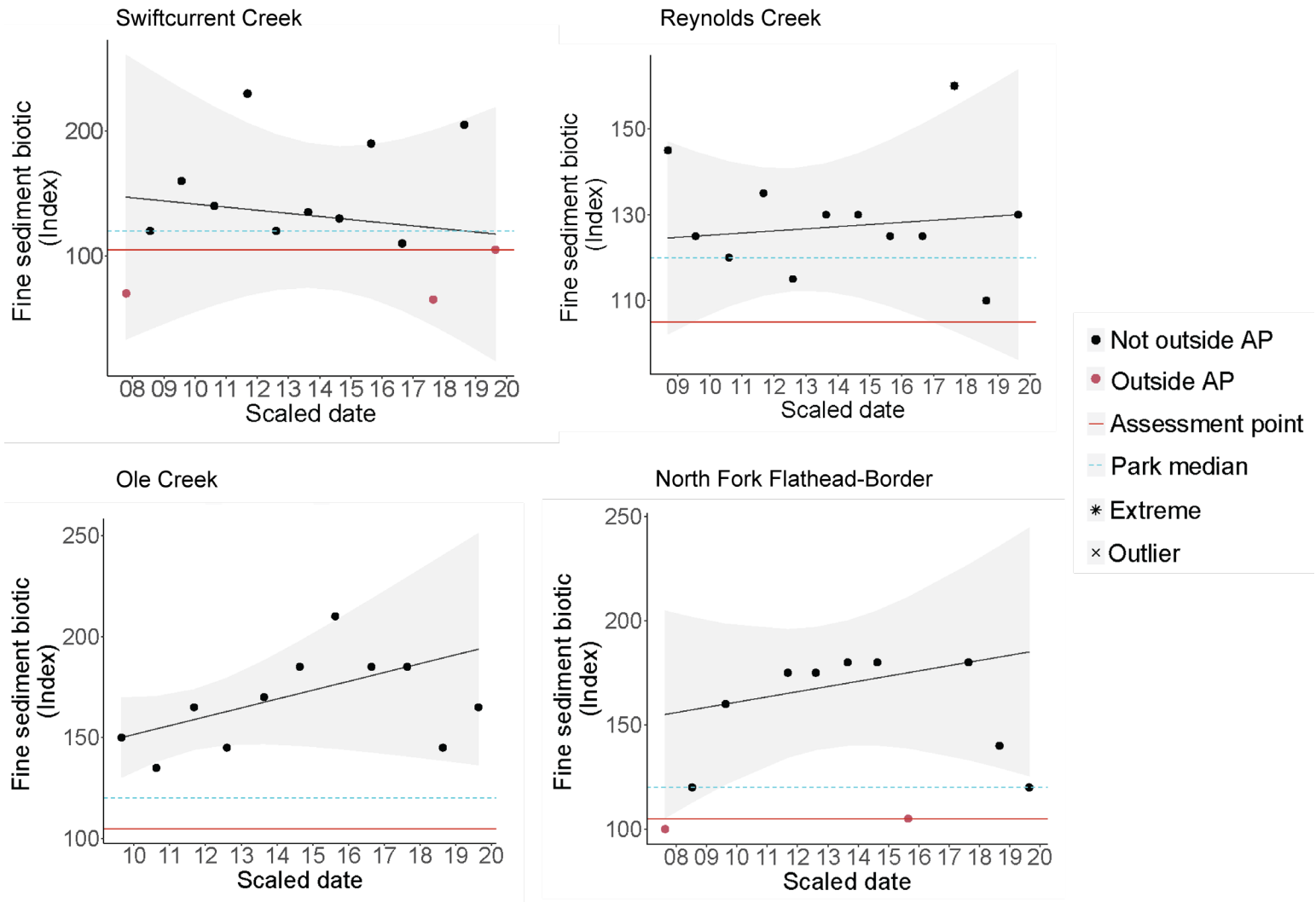


Figure 63. Fine Sediment Biotic Index in Glacier National Park from 2007 to 2019. (Top left) Swiftcurrent Creek, (top right) Reynolds Creek, (bottom left) Ole Creek, and (bottom right) North Fork Flathead—Border. As available, red horizontal lines show a non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (no significant trends at these four sites) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

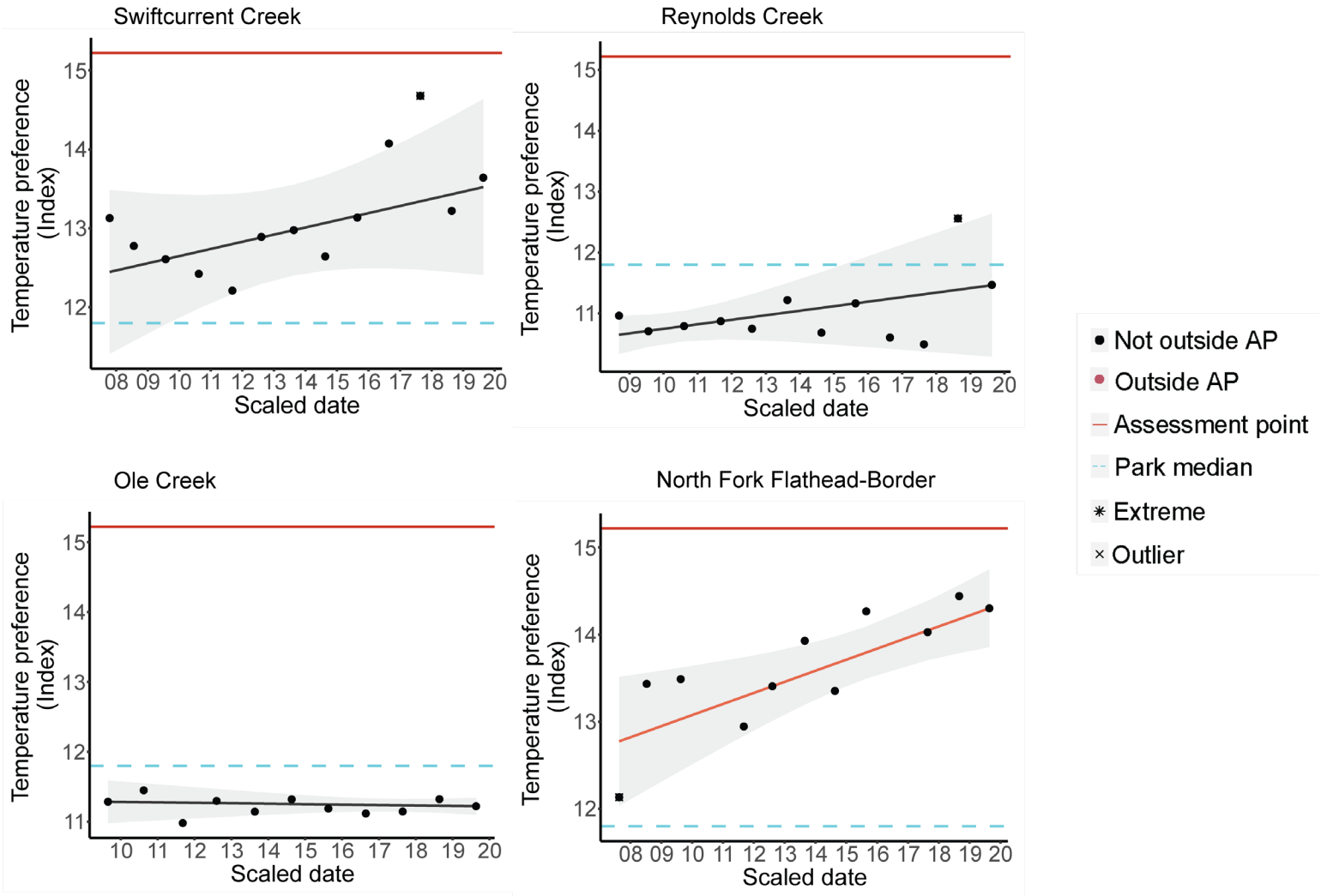


Figure 64. Temperature Preference Index in Glacier National Park from 2007 to 2019. (Top left) Swiftcurrent Creek, (top right) Reynolds Creek, (bottom left) Ole Creek, and (bottom right) North Fork Flathead—Border. As available, red horizontal lines show a non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (significant trend at North Fork Flathead—Border) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

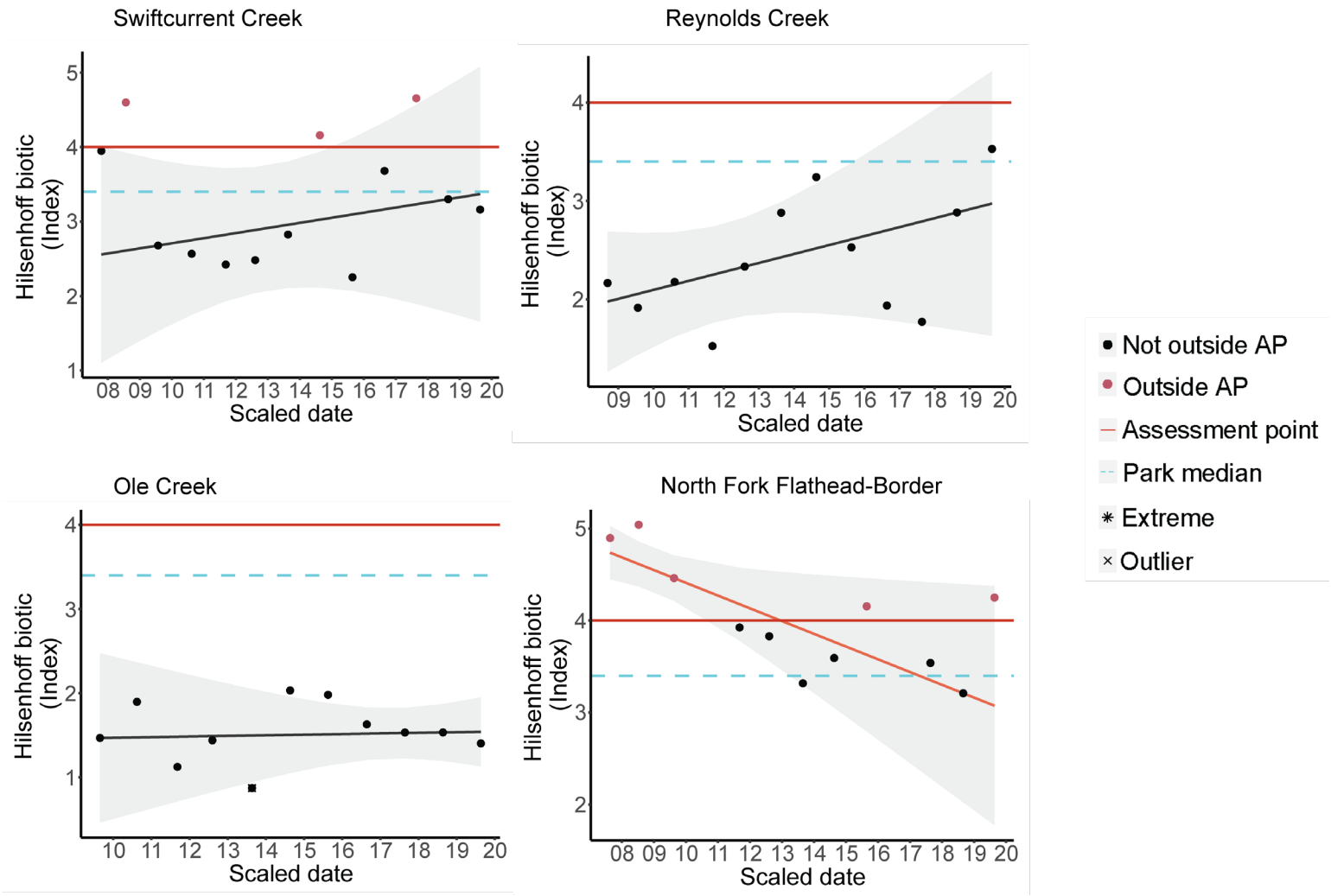


Figure 65. Hilsenhoff Biotic Index (HBI) in Glacier National Park from 2007 to 2019. (Top left) Swiftcurrent Creek, (top right) Reynolds Creek, (bottom left) Ole Creek, and (bottom right) North Fork Flathead—Border. As available, red horizontal lines show a non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (significant trend at North Fork Flathead—Border) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

3.5.4.5 Condition

Several macroinvertebrate stressor-specific metric results were outside a reference range suggested by our assessment points (Table 27). FSBI at Swiftcurrent Creek was below the assessment point of 15.4%, and at the North Fork Flathead—Border site, macroinvertebrate assemblage composition suggested there were excess fines at the site 9.1% of the time over the period of record (Figure 63). Most of the potentially problematic values at Swiftcurrent Creek occur late in the period of record concurrent with or after the beaver dam was in place. Reduced flow velocity caused by the dam may have elevated fine sediments. It is also possible that the response is to the higher levels of visitor use of the site. The unresolved cause behind higher fine sediments was also our general interpretation of measured sediment data as presented in the chemistry and habitat sections above. TPI values approached but did not exceed the assessment point at most sites. Values for the North Fork Flathead—Border site came close to exceeding the assessment point. Finally, HBI was above the Montana DEQ criteria we use as an assessment point up to 50% of the time at all sites except Reynolds Creek and Ole Creek. Moreover, three of our four sites had nonsignificant positive trend coefficients for HBI. Should more data lead to significant trend models, HBI might exceed its criteria more often in the future. The HBI index is part of the Montana DEQ integrated nutrient assessment we develop in the chemistry section above, and these elevated values are taken into consideration. However, HBI values above the criterion cannot alone lead to a designation of a site as nutrient impaired. If this is coupled with other indicators of nutrient issues, a site may be so designated. The required combination of values does not occur at any sentinel site over our period of record.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites. Median FSBI, TPI, and HBI values at these sites were largely similar to other sites. However, 50% of the readings at North Fork Flathead—Glacier Rim and the Middle Fork Flathead were above the assessment point for HBI, and at the Middle Fork Flathead for TPI. There is some recreational use of these sites and limited development in their watershed, yet with our limited sample size to date at these sites (only 2 data points), any firm interpretations are not possible.

3.5.4.6 Discussion and management applications

Fine Sediments

Increased levels of fine sediment beyond that which occurs naturally in a system can influence stream biota by reducing photosynthesis and thus production, physically abrading algae, or preventing attachment of autotrophs to substrate surfaces (Brookes 1986; Relyea et al. 2012). Macroinvertebrates experience lowered respiration via fines blocking gill surfaces (Figure 66), as well as reduced efficiency of filter feeding, visual predation, and grazing, with food supplies either buried under sediments or diluted by increased inorganic sediment loads. Interstitial spaces between larger sediments are important because many macroinvertebrates can be smothered in fines (the conceptual definition of our embeddedness indicator, see Figure 52).



Figure 66. Mayfly nymph in Glacier National Park with a layer of fine sediment covering gills and other body parts. Image courtesy J. Giersch, USGS.

The sediment specific FSBI suggests a small but consistent negative macroinvertebrate response to fines at Swiftcurrent Creek and to a lesser degree at the North Fork Flathead—Border. The Swiftcurrent Creek assemblage often had taxa like *Zapada oregonensis* or *Heterlimnius corpulentus*, rated in Relyea et al. (2012) as only slightly fine sediment sensitive. They are in the least sensitive category and thus better able to persist in streams where fines are more prevalent.

All three of our habitat sediment metrics also showed consistent (e.g., no trend) but higher levels of fine sediments at Swiftcurrent Creek (see physical habitat section above). Relative bed stability was also on the fining side at Swiftcurrent Creek, but with a (non-significant) decreasing value across the period of record. In other words, bed stability at the site was not becoming significantly more fine sediment biased, but it was pointed in that direction. Exploratory data analyses suggests that FSBI is correlated with relative bed stability across all our sentinel sites; higher FSBI, or more sediment sensitive taxa, occur with higher relative bed stability, or more stable and larger sediments. Interestingly, we also see an interpretable relationship between FSBI and HDI, with lower FSBI scores (more sensitive tolerant taxa) in sites with higher general human disturbance. While clearly this needs more careful treatment, it is suggestive of a human disturbance signal driving a sediment response in macroinvertebrates.

The biological response seen in FSBI at Swiftcurrent Creek, and to a lesser degree at the North Fork Flathead—Border, mirrors patterns in sediment data. While no trends were significant, several results are suggestive. A decreasing FSBI indicates increased levels of fine sediments, as we see in the habitat data. We interpreted higher fine sediment habitat data as likely due to the natural effects of

the beaver dam at Swiftcurrent Creek for a few years, but also potentially because of the relatively high visitor use and adjacent campground (see Figure 37). However, only 15% of the FSBI data points were in a non-reference condition, and we feel this mitigates an interpretation of possible sediment issue at Swiftcurrent Creek from habitat data alone. Additional data and monitoring are warranted given these patterns.

Temperature

As noted throughout this report, stream water temperature is a key driver of the ecology of the park's streams. Much of the water temperature in the park is warming. Water temperatures influence growth, feeding, and metabolic rates; emergence; fecundity; and behavior and survival of macroinvertebrates (Dallas and Rivers-Moore 2012). These responses to water temperature integrate across individuals to a species and, ultimately, an assemblage level response, as estimated by the Temperature Preference Index (TPI). Yet water temperature is difficult to measure at all sites. An index like the TPI may be an efficient way to accomplish this. The TPI provides a useful bioassay of temperature dynamics in a stream—in a sense the TPI treats the macroinvertebrate assemblage as a natural temperature logger that integrates highly variable water temperatures experienced across the life cycles of the taxa in a stream. Taxa like *Diphetor hageni* and *Brachycentrus occidentalis* (Figure 67), with temperature optima in Brandt (2001) of 19 and 17 °C, respectively, are likely to persist across their life cycle in streams with these high temperatures. Taxa like these are typical of the increasingly warm water tolerant macroinvertebrate assemblages in some Glacier National Park sentinel sites. While these species have likely been in Glacier's streams for a long time, increases in bioindicators like the TPI suggest that they may be becoming more common at select sites in the park. At the North Fork Flathead—Border site, this was a strongly significant increasing trend—one of the strongest in our entire dataset. Models of continuous logger-based temperature also show increasing water temperature, often related to stream flow, soil moisture, and evapotranspiration (see Appendix C) These concordant patterns provide additional weight of evidence to the story of the park's warming waters.



Figure 67. Warm water tolerant *Diphetor hageni* (adult) and *Brachycentrus occidentalis* typical of Glacier National Park. Images courtesy J. Giersch, USGS.

Finally, exploratory work indicates a rich array of connections between TPI and site level water chemistry, habitat, and other biological measures. For example, TPI increases with specific

conductance, suggesting more high temperature tolerant macroinvertebrates persist in less dilute waters that are often in lower elevation larger rivers (that are relatively warmer). We also see more high temperature tolerant macroinvertebrates in systems with higher pool density. The slack water of pools is typically warmer. These patterns need further exploration, but they support a structural, wide ranging role of water temperature in the park's stream ecology and suggest future models that can capture these relationships in a predictive context.

Nutrients

Nutrient levels are another stressor to Glacier National Park sentinel stream sites featured throughout this report. Nutrients are a primary focus of Montana DEQ regulatory work, and there is a long history of research on the trophic status of aquatic systems. However, the relationship is complex, and it is difficult to treat "nutrients" as a broad stressor type given complex interactions between nitrogen and phosphorous limitation, the importance of bio-available forms of nitrogen and phosphorous, and threshold effects. Part of the value of a bioindicator like HBI is its ability to integrate across many of these complexities. The HBI is featured in several long-term monitoring efforts from the US EPA, the USGS, and multiple states. It is often a component metric in multimetric indices. Species diagnostic of nutrient enrichment (high HBI scores) and increasingly more common at select sentinel sites include the tolerant *Diphetor hageni* (also tolerant of warmer water, see Figure 67) and the nutrient sensitive *Epeorus grandis* (Figure 68).



Figure 68. *Epeorus grandis* in a Glacier National Park stream. This mayfly is sensitive to higher nutrient concentrations. Image courtesy J. Giersch, USGS.

Of the three macroinvertebrate stressor specific indicators, HBI was often above its assessment point, especially at the Swiftcurrent Creek and North Fork Flathead—Border sites. However, the North

Fork Flathead—Border HBI was also one of the few significantly decreasing trends in our bioindicator data, with levels dropping across our period of record. The response at this site was still variable, with individual values above the assessment point and in non-reference condition near the end of data even as the overall trend decreased. We also see higher levels of total phosphorous at the North Fork Flathead—Border site in our water chemistry data, but not as high as with HBI, suggesting some amplification of a nutrient effect in the biological response. Using the Montana DEQ integrated nutrient assessment protocol, this combination does not suggest a nutrient impairment.

There are regional patterns in nutrients, with increases in phosphorous in remote wilderness watersheds like most of Glacier National Park. These watersheds may be more influenced by elevated movement of naturally occurring phosphorus via climate driven increases or changes in the timing of runoff increase contribution. Schweiger et al. (2018) showed a weak but positive relationship between HBI and both total nitrogen and phosphorous in the 2007–2009 baseline Glacier survey but, as in our sentinel site results, non-reference HBI results suggested higher nutrient response than the measured levels of total nitrogen or phosphorous (or any other nutrients). This again suggests an amplified biological response in our baseline. In both cases, it is unclear if this was because bioassessment reflected true effects of elevated nutrients or issues within our analytical processes. Given a lack of known human sources for nutrients in the park, Schweiger et al. (2018) concluded that, either way, there was not much cause for concern, and we expect that is still the case over time at our sentinel sites—although it does warrant more research and some caution.

Finally, as the with TPI, exploratory work indicates a rich array of connections between HBI and site level water chemistry, habitat, and other biological measures. For example, HBI increases with HDI, offering some correlative evidence that the patterns we present above in non-reference HBI and nutrient levels may have at least an indirect and general human disturbance connection. In addition, we see a stronger positive connection between HBI and total phosphorous than with total nitrogen. This suggests phosphorous may be more limiting than nitrogen, matching regional patterns suggested by Stoddard et al. (2016). Finally, we see a response to pool density. However, the pattern may be counterintuitive, with more tolerant to nutrient macroinvertebrates in sites with higher pool density. We might expect more nutrient tolerant taxa in sites with more slack water, although this is likely complex.

3.5.5 Diatoms

The following sections present results and interpretation of select diatom bioassessment metrics at sentinel sites sampled in 2007–2019. Additional results, specifically an update to the synthetic Glacier Multimetric Index for diatoms, is currently under production and will be presented in a future publication. Here we focus on patterns over time in stressor-specific metrics. Schweiger et al. (2018) presents similar information for the population of streams in the park.

3.5.5.1 Background and relevancy

The State of Montana has been using diatoms to assess water chemistry in streams and rivers since the 1970s (Bahls et al. 2008). Early efforts were based on Shannon diversity (Weber 1973), autecological tendencies compiled by Lowe (1974), and the pollution tolerance classes of Lange-

Bertalot (1979). The Glacier Diatom Multimetric Index presented in Schweiger et al. (2018) applied bioassessment to the diatom communities in the park, extending this earlier work by the State of Montana and its partners. However, these methods do not evaluate specific pollutants. Therefore, we applied two models that directly link diatom response to important specific stressors in Glacier—nutrients and sediment (Table 28).

Table 28. Select diatom indicators at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance (with the value bolded if at or below 0.10. Percent in exceedance is the percentage of results above (and/or below) the given assessment point(s) (AP). AP types are all regulatory criteria. The diatom increaser indices increase as condition in a stream deteriorates, and values above or equal to the AP are in a non-reference condition. AP were derived from State of Montana aquatic life support standards and are the percent relative abundance of increaser taxa where the probability of impairment is at or greater than 0.50.

Variable	Site	Trend in partial residual	Trend (p)	Percent in exceedance of AP	AP	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Northern/ Canadian Rockies nutrient increasers	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	0.1518	8/25/2007	8/20/2019	4	0.053	0.045	0.059	0.043	0.025	0.121
	North Fork Flathead—Border	−0.0023	0.607	0.0	0.1518	8/20/2007	8/22/2019	11	0.053	0.067	0.078	0.039	0.04	0.149
	Swiftcurrent Creek	−0.0022	0.159	0.0	0.1518	10/18/2007	8/22/2019	13	0.053	0.036	0.037	0.020	0.007	0.082
	Reynolds Creek	−0.0033	0.456	8.3	0.1518	9/11/2008	8/20/2019	12	0.053	0.059	0.067	0.049	0	0.156
	Ole Creek	−0.0012	0.903	9.1	0.1518	8/27/2009	8/21/2019	11	0.053	0.033	0.071	0.094	0.007	0.332
	Middle Fork Flathead	– [a]	– [a]	0.0	0.1518	8/28/2018	8/19/2019	2	0.053	0.013	0.013	0.000	0.013	0.013
Northern/ Canadian Rockies sediment increasers	North Fork Flathead—Glacier Rim	– [a]	– [a]	0.0	0.1848	8/25/2007	8/20/2019	4	0.077	0.062	0.079	0.040	0.053	0.138
	North Fork Flathead—Border	−0.0056	0.317	9.1	0.1848	8/20/2007	8/22/2019	11	0.077	0.113	0.124	0.051	0.056	0.233
	Swiftcurrent Creek	−0.0046	**0.078** [b]	0.0	0.1848	10/18/2007	8/22/2019	13	0.077	0.066	0.061	0.028	0.018	0.114
	Reynolds Creek	0.0027	0.692	8.3	0.1848	9/11/2008	8/20/2019	12	0.077	0.099	0.099	0.051	0.018	0.188
	Ole Creek	−0.0045	0.631	9.1	0.1848	8/27/2009	8/21/2019	11	0.077	0.097	0.108	0.089	0.012	0.345
	Middle Fork Flathead	– [a]	– [a]	0.0	0.1848	8/28/2018	8/19/2019	2	0.077	0.017	0.017	0.004	0.015	0.02

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] Trend is significant ($p \leq 0.10$).

Nutrient and sediments: Increaser models

Beginning in the early 2000s, Teply and Bahls (2005, 2006) and Teply (2010a, b) developed diatom based bioassessment "increaser" models. These models identify diatom taxa (Figure 69) that exist in detectable amounts in stream types defined by high levels of fine sediment or nutrients. Taxa unique to these cause-specific lists are considered ecological specialists that are tolerant of excess fine sediments or nutrients. Increaser models are based on empirical observations across many streams classified by ecoregion and type of impairment. Models identify taxa based on increases in average abundance in the presence of a stressor and then use discriminant functions to estimate the probability of impairment given these relative abundances. Because the models are defined for ecoregions, increaser taxa are regionally specific, reflecting meaningful differences in geology, hydrology, and natural variability taxa assemblage composition. Increaser models are well suited to the ROMN long term monitoring protocol and our Glacier diatom data. Nearly 100% of the species on each list occurred in our samples. This suggests the models are well calibrated to the park's streams and rivers.

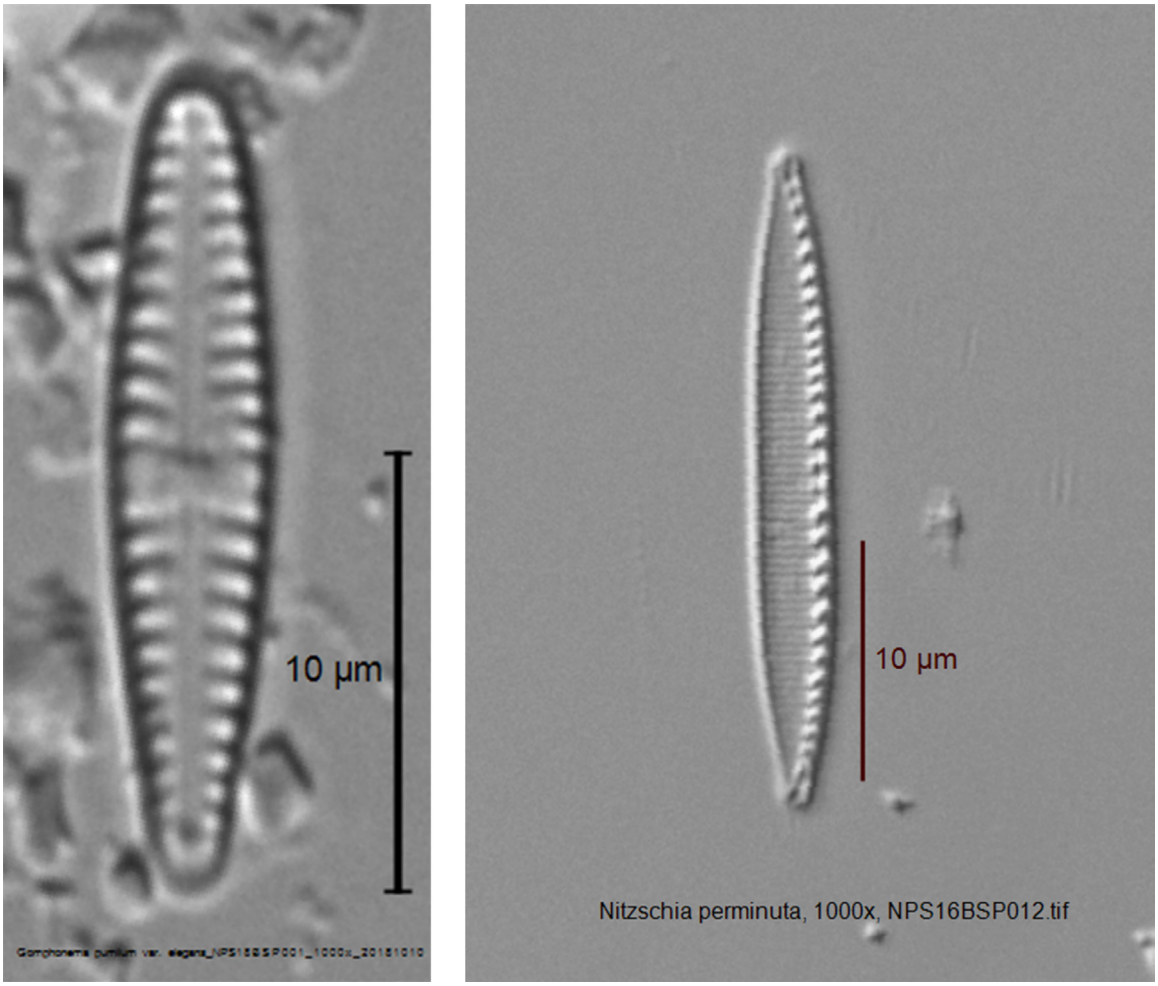


Figure 69. Sediment and nutrient increaser diatoms *Gomphonema pumilum* from the North Fork Flathead River border site and *Nitzschia perminuta* from Swiftcurrent Creek. (NPS)

3.5.5.2 Assessment points

Montana DEQ (Teply 2010a, b; Suplee and Sada de Suplee 2016) developed a set of criteria for increaser models in support of regulatory decisions that are relevant for use in Glacier National Park as assessment points (again, any authority related to them rests with Montana DEQ and its interaction with Glacier National Park resource management).

3.5.5.3 Parkwide 2007–2009 baseline results

Schweiger et al. (2018) found that 3% and 5% of the streams in the park had diatom assemblages suggesting non-reference sediment and nutrient levels, respectively. This was roughly equivalent to the amount of stream length with higher levels of fine sediments or elevated nutrients, suggesting a roughly equal biological response to these stressors. Most sites with higher sediment and nutrient levels were in or near the front country, suggesting potential human sources for fine sediments and nutrients, such as roads or other visitor services infrastructure.

3.5.5.4 Trend

There were no significant or marginally significant trends in nutrient increasers at any sentinel site (Table 28, Figure 70). However, the directions of slopes were consistently negative across sites. The model at Swiftcurrent Creek approached significance, suggesting a more pronounced decrease in nutrient increaser data over time at Swiftcurrent Creek over our period of record.

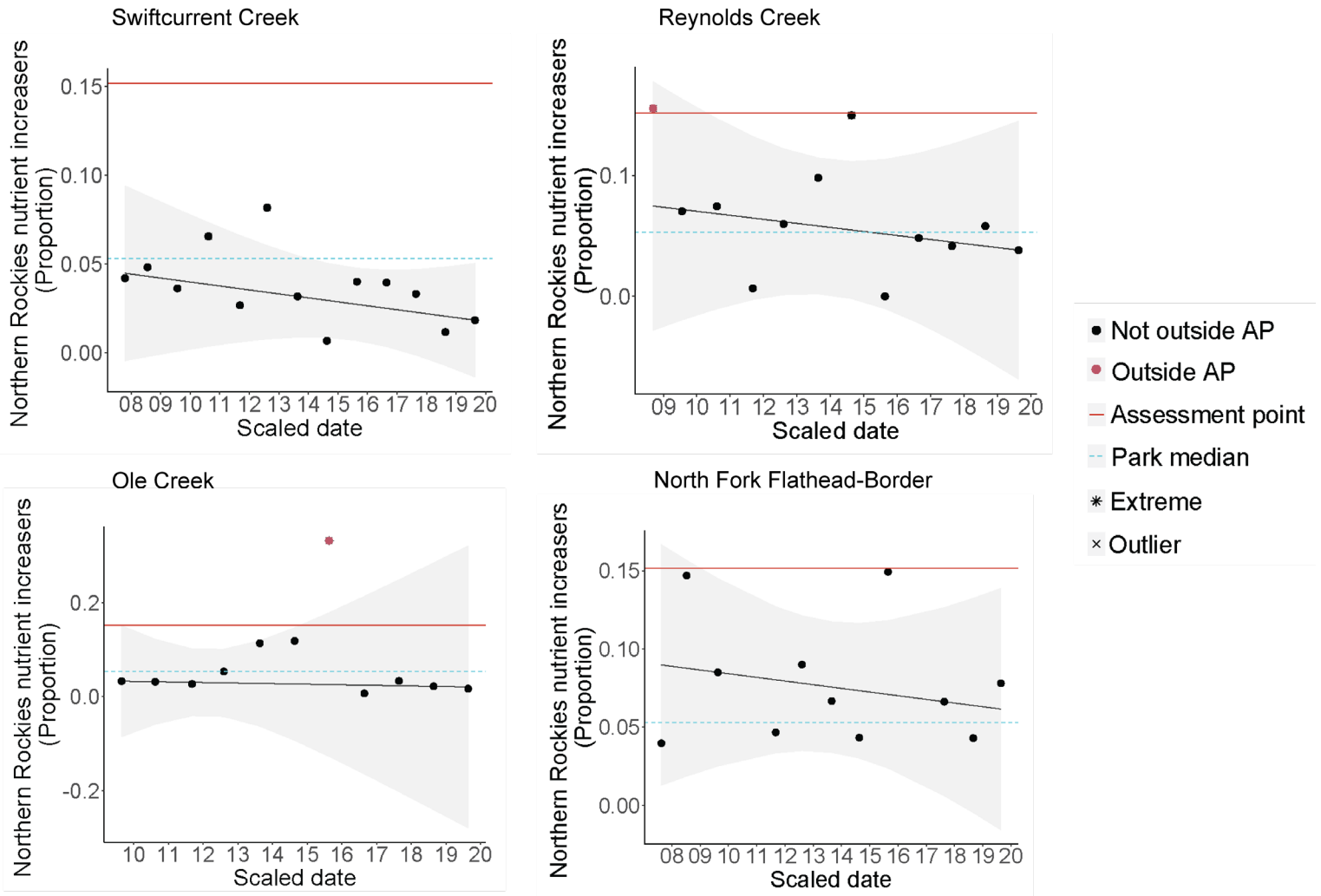


Figure 70. Northern/Canadian Rockies nutrient increasers index in Glacier National Park from 2007 to 2019. (Top left) Swiftcurrent Creek, (top right) Reynolds Creek, (bottom left) Ole Creek, and (bottom right) North Fork Flathead—Border. As available, red horizontal lines show a non-reference assessment point and a dashed blue line shows the park-scale 2007–2009 median. The fitted lines (no significant trends at these four sites) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

There was one significant trend in sediment increasers: a decrease over our period of record at Swiftcurrent Creek (Table 28, Figure 71). Other sites had mixed responses in sediment increasers, none of which were significant.

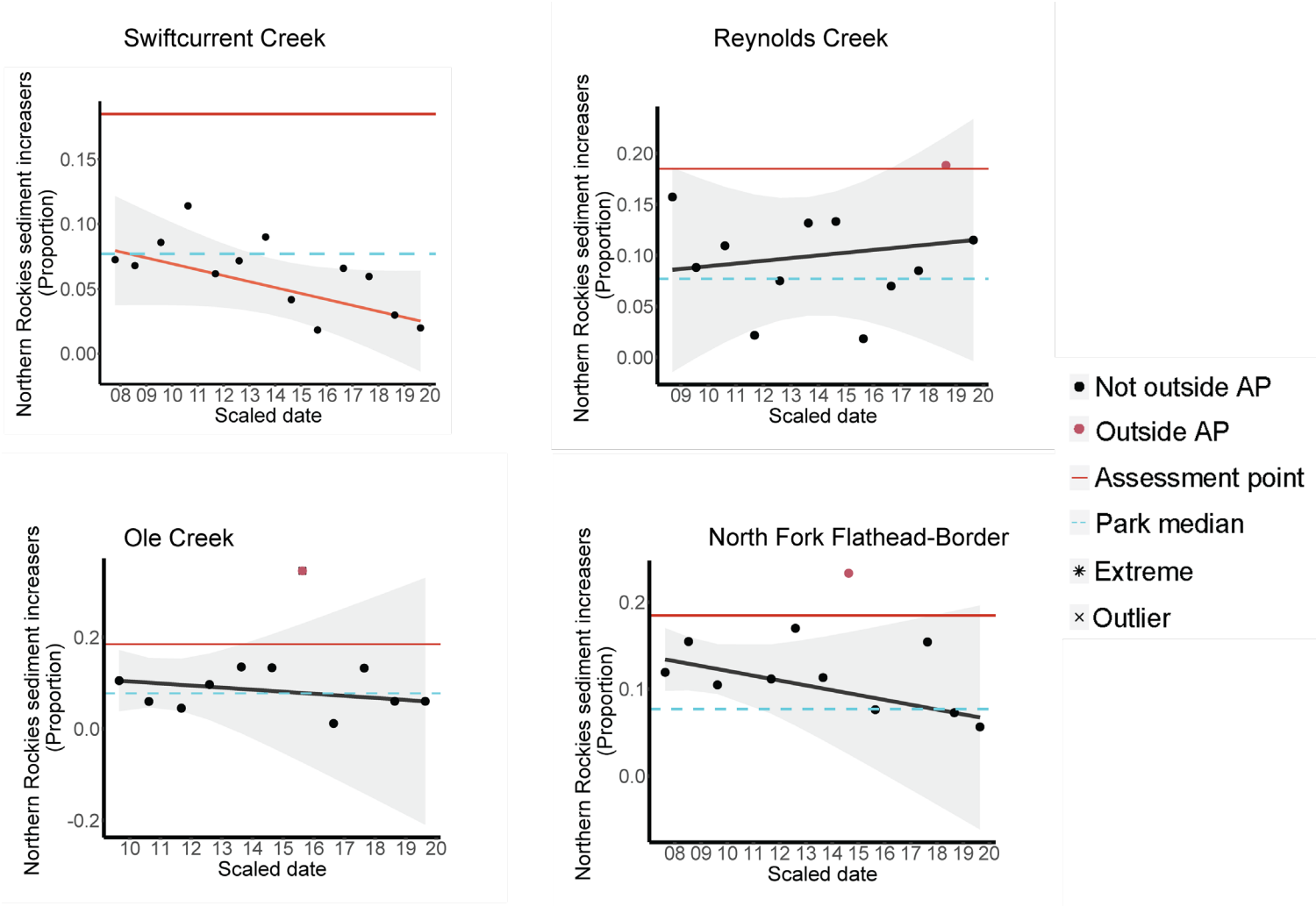


Figure 71. Northern/Canadian Rockies sediment increasers index in Glacier National Park from 2007 to 2019. (Top left) Swiftcurrent Creek, (top right) Reynolds Creek, (bottom left) Ole Creek, and (bottom right) North Fork Flathead—Border. As available, red horizontal lines show a non-reference assessment point and a dotted blue line shows the park-scale 2007–2009 median. The fitted lines (significant trend at Swiftcurrent Creek) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

3.5.5.5 Condition

Several diatom increaser metric results were outside a reference range suggested by our assessment point (Table 28). Nutrient increasers were above the assessment point from 8% to 9% of the time at North Fork Flathead—Border, Reynolds Creek, and Ole Creek (Figure 70). Sediment increasers were above the assessment point 8% and 9% of the time at Reynolds Creek and Ole Creek, respectively (Figure 71). The diatom response at the North Fork Flathead—Border and Reynolds Creek for both nutrients and sediments was nearly above the assessment point in a second event, suggesting that, at these sites, select sample events were slightly more reflective of increased sediments or nutrients. However, in all cases, these percent exceedances represent single events within each site's period of record, and in nearly all cases, any trend was away from the assessment point.

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites. Median diatom increaser metric results at North Fork Flathead—Glacier Rim are like other sites; however, values at the Middle Fork Flathead site were quite low (suggesting less of a diatom response to sediments or nutrients). No results were above the assessment point for either increaser model at either site, but with our limited sample size to date at these sites (only two data points), any firm interpretations are difficult.

3.5.5.6 Discussion and management applications

Excess sediments and nutrients are serious issues in Montana, with about half of the impaired streams in Montana impacted by sediment (Teply 2010a, b). While most of Glacier National Park's aquatic systems do not face the same degree of localized human sediment or nutrient sources as many streams outside the park, select front country systems can be impacted by these stressors. Large-scale drivers, including climate change, can also interact with sources of these stressors and enhance their impact.

Nutrients

The relationships between diatoms and nutrients have long been key in stream monitoring. For example, excess nutrients alter diatom growth rates and can lead to depleted dissolved oxygen levels (Rhodes et al. 2017). Diatoms also have strong impacts on nutrients—blooms are often associated with depleted nutrients in a system, which can have a cascade of effects on other aquatic taxa. For example, reduced nutrients can alter food webs, deplete oxygen levels, and reduce light penetration levels in the water column.

The directions of change in diatom nutrient increasers was negative across all sites. This suggests a consistent reduced biological response in diatoms to nutrients. Moreover, the small percentage of nutrient increaser results suggesting impairment also supports a more muted nutrient signal in diatoms. Perhaps one exception to this is at the North Fork Flathead—Border site, where two diatom events were just below the impairment criterion. The macroinvertebrate HBI is also often above its threshold at this site, where we also measured higher levels of total phosphorous. Nevertheless, the integrated state method, where all this information is put together, does not suggest nutrient impairment over our period of record.

Exploratory work suggests several possible connections between nutrient increasers, water chemistry, site level habitat, and other biological measures. For example, we see a positive relationship with stream flow, much like the role of flow in the trend models of total phosphorous or nitrogen (see Figure 40). Phosphorous consistently has stronger and positive relationships with nutrient increasers, while nitrogen is negative. This suggests that nutrient sensitive diatoms are likely limited by phosphorous (i.e., when it has a higher concentration, these species respond with higher relative abundances). Phosphorous limitation of aquatic systems and its impact on diatoms (Elser et al. 1990; Seip 1994) is well documented. This has become an important part of how regional scale patterns in nutrients are understood (i.e., Stoddard et al. 2016) and will likely be important in our continued work on nutrients in Glacier National Park.

Sediment

The potential impact of increased levels of fine sediment on diatoms (Jones et al. 2014) mirror many of those described for macroinvertebrates. Layers of fine sediment bury diatoms, many of which are not motile (much like a rooted plant). Several diatom taxa, however, are unique in that they can move and relocate away from deposited fines—these are the diatom taxa that are in the sediment increaser index as this motility allows them to better persist in the presence of fines.

The significant negative trend in diatom sediment increasers at Swiftcurrent Creek suggested a decrease in fine sediments at the site over our period of record. This was also seen in the (nonsignificant) decrease in macroinvertebrate sediment response at the site. However, unlike with macroinvertebrates, no diatom sediment increaser values were above the state criteria of impairment. Trends in our actual sediment data at Swiftcurrent Creek (see Table 22) were generally flat, yet there were many results above the assessment point, suggesting some concern. These patterns suggest a complex response, but in general, may indicate the biological response was more muted or that there may be a lag in the relationship. We feel additional data and monitoring is warranted.

Finally, exploratory work indicates a rich array of connections between sediment increasers and site level habitat and other biological measures. For example, sediment increasers and percent fines have an intuitive relationship, with a higher probability of impairment where we also see more fine sediments. We also see more sediment increasers in systems with higher stream flow. This suggests a stronger response in larger rivers. These will need further exploration, but they suggest future models that can capture these relationships in a predictive context.

Metals

Schweiger et al. (2018) included a third increaser model for metals (Teply and Bahls 2005, 2006). This was based on a similar approach as summarized above but developed for a much broader mountain bioregion (hence “Mountain metals increasers”). We applied the tool to the 2007–2009 baseline data, with many caveats, as the overall classification accuracy (in its developmental work) was often below 50% and false positive rates were mostly well over 30%, which suggested that the method was not even marginally reliable for regulatory use in stream impairment determinations. For this report we have elected to omit these results. We recognize this is less than ideal, given the possible issues with metals we see especially at Reynolds Creek, but with a real risk of false

positives/negatives, it seems prudent. Future work should focus on biological response to metals in our Glacier National Park sentinel sites.

3.5.6 Aquatic invasive species

3.5.6.1 Rock snot

Didymosphenia geminata (commonly known as rock snot or didymo) is a diatom native to mountain aquatic habitats of North America and Europe (Blanco and Ector 2009). Didymo can form extensive mats (Figure 72), which can be several centimeters thick and up to 20 km (12 mi) in length (Blanco and Ector 2009). Larger blooms can inhibit growth of other algal species, change the composition of aquatic communities, decrease the amount of suitable spawning habitat for fish, and cause changes in stream chemistry (Spaulding and Elwell 2007). Blooms of didymo also greatly decrease the aesthetic appeal of streams—an important consideration for a tourist destination like Glacier National Park. In recent years, didymo has expanded into lower elevations, latitudes, and new regions of the globe (Kumar et al. 2009). In Montana, didymo was first reported in 1929 at Flathead Lake (Prescott and Dillard 1979) but it has likely been present in the northern Rockies since at least the end of the last ice age, about 10,000 years ago (Bahls 2007).



Figure 72. *Didymosphenia geminata* in Swiftcurrent Creek, Glacier National Park, 2008. NPS / BILLY SCHWEIGER.

We present trend models and comparisons to ecoregional assessment points of a didymo index based on lab samples and a visual field data-based Kilroy bloom index (Kilroy and Unwin 2011) at sentinel sites from 2007 to 2019. Our didymo index is the total count of didymo cells (known as "valves") from whole slide scans. Given the size of didymo valve, the usual subsampling of slides produced by the Rhithron lab from each field sample is not a valid way to document didymo. Analysts therefore scan the whole slide and count all didymo on each slide. Because each slide is searched with consistent effort and each field sample is produced in the same way, we do not need to scale these counts, but the results are still treated as a unitless index for didymo abundance. Unlike in Schweiger et al. (2011) and several other similar treatments (i.e., Spaulding and Elwell 2007; Jackson et al. 2016), we do not include covariates in our trend models, instead focusing on simpler estimates of trend. Future work should continue the analyses of structural relationships that began in Schweiger et al. (2011).

Assessment points

There are no management, regulatory, or known ecological assessment points for our didymo or the Kilroy indices. For these two metrics, reference sites were limited to a subset of 38 sites in the park with minor or negligible human disturbance (see map of ecoregional assessment sites in Appendix A; Schweiger et al. 2018; Suplee et al. 2005). The assessment point was defined at the 95th percentile from didymo index and Kilroy bloom index distributions (Table 29). Values above or equal to this assessment point are in a non-reference condition. This assessment point assumes the 2007–2009 period provides a useful baseline for these indicators. An ecological threshold would be a preferred assessment point type given likely complex interactions and effects of didymo on other aspects of streams in the park. Future work should seek to develop alternate and/or ecological thresholds to be used as assessment points for didymo indicators.

Table 29. Select *Didymosphenia geminata* (didymo) metrics at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. The Trend field gives the coefficient from a quantile regression for date with the Trend (p) field its significance. Assessment point (AP) types are all ecoregional. We use ecoregional assessment points from > 95th percentile value from 38 ecoregional reference sites. Percent outside values are not reported because all were zero.

Metric	Site	Trend in partial residual	Trend (p)	AP	Start date	End date	Sample size	2007–2009 park scale median	Median	Mean	Standard deviation	Min	Max
Didymo index	North Fork Flathead—Glacier Rim	– [a]	– [a]	893	8/25/2007	8/20/2019	4	38.650	79.500	79.000	59.738	7	150
	North Fork Flathead—Border	−0.8741	0.222	893	8/20/2007	8/22/2019	11	38.650	1.000	5.182	6.478	0	20
	Swiftcurrent Creek	−2.1055	0.689	893	10/18/2007	8/22/2019	13	38.650	25.000	44.077	56.436	1	195
	Reynolds Creek	– [b]	– [b]	893	9/11/2008	8/20/2019	12	38.650	0.000	0.417	0.900	0	3
	Ole Creek	−0.3328	0.987	893	8/27/2009	8/21/2019	11	38.650	3.000	77.818	195.999	0	645
	Middle Fork Flathead	– [a]	– [a]	893	8/28/2018	8/19/2019	2	38.650	13.000	13.000	8.485	7	19
Kilroy index	North Fork Flathead—Glacier Rim	– [a]	– [a]	72.8	8/25/2007	8/20/2019	4	0.000	1.741	2.399	2.167	0.637	4.818
	North Fork Flathead—Border	– [b]	– [b]	72.8	8/20/2007	8/22/2019	11	0.000	0.000	0.000	0.000	0	0
	Swiftcurrent Creek	1.15	0.582	72.8	10/18/2007	8/22/2019	13	0.000	4.655	14.289	17.478	0	50.459
	Reynolds Creek	– [b]	– [b]	72.8	9/11/2008	8/20/2019	12	0.000	0.000	0.000	0.000	0	0
	Ole Creek	0.0465	0.8983	72.8	8/27/2009	8/21/2019	11	0.000	0.182	1.443	3.044	0	9.227
	Middle Fork Flathead	– [a]	– [a]	72.8	8/28/2018	8/19/2019	2	0.000	2.273	2.273	2.314	0.636	3.909

[a] "–" (en-dash) indicates that the period of record at North Fork Flathead—Glacier Rim and Middle Fork Flathead does not support trend estimation.

[b] "–" (en-dash) indicates that Reynolds Creek and the North Fork Flathead—Border had insufficient didymo or Kilroy results and trend models were not possible.

Parkwide 2007–2009 baseline results

Results presented in Schweiger et al. (2011) from the 2007–2009 survey showed that didymo was widespread, occurring in 64% of the park's streams. Models suggested that didymo was positively related to summer water temperatures, the proportion of cobble substrate, and Julian date (indicating that didymo abundance increased within years during our sample period). Conversely, didymo abundance was negatively related to distance from lakes (e.g., with higher abundances near lake outlets), total nitrogen concentration, and specific conductance. It was not connected to human use of streams or their riparian corridors. Of 11 sites revisited in 2007–2009, didymo abundance increased at eight and decreased at only one. Abundance also varied spatially, with the highest abundances of didymo on the west side of the Continental Divide.

Schweiger et al. (2011, 2018) concluded that didymo was potentially an emergent issue for the park, needing ongoing monitoring. Important climate change relevant predictors of elevated didymo abundance included higher stream temperature and stable base flows (using a proxy of distance from upstream lakes). These suggested that didymo was likely to respond to changing climate. Additional work is needed to define numeric thresholds in drivers like water temperature and nutrient concentrations that are specific to Glacier.

Trend

There were no significant or marginally significant trends in the didymo or Kilroy indices at any sentinel site (Table 29). Didymo was detected at Reynolds Creek only in 2010, 2012 and 2017, and only as a single or a few (in 2012) valves and never as a bloom (Kilroy index). We suspect may be due to contamination of sampling gear. At other sites, the directions of slopes (Figure 73) for the didymo index were all negative, suggesting that didymo (non-significantly) decreased at these sites over our period of record. At Swiftcurrent Creek and Ole Creek, trends in the Kilroy index were both positive but nonsignificant (Figure 74). There were insufficient data to estimate a trend in the didymo or Kilroy index at Reynolds Creek and the North Fork Flathead—Border site.

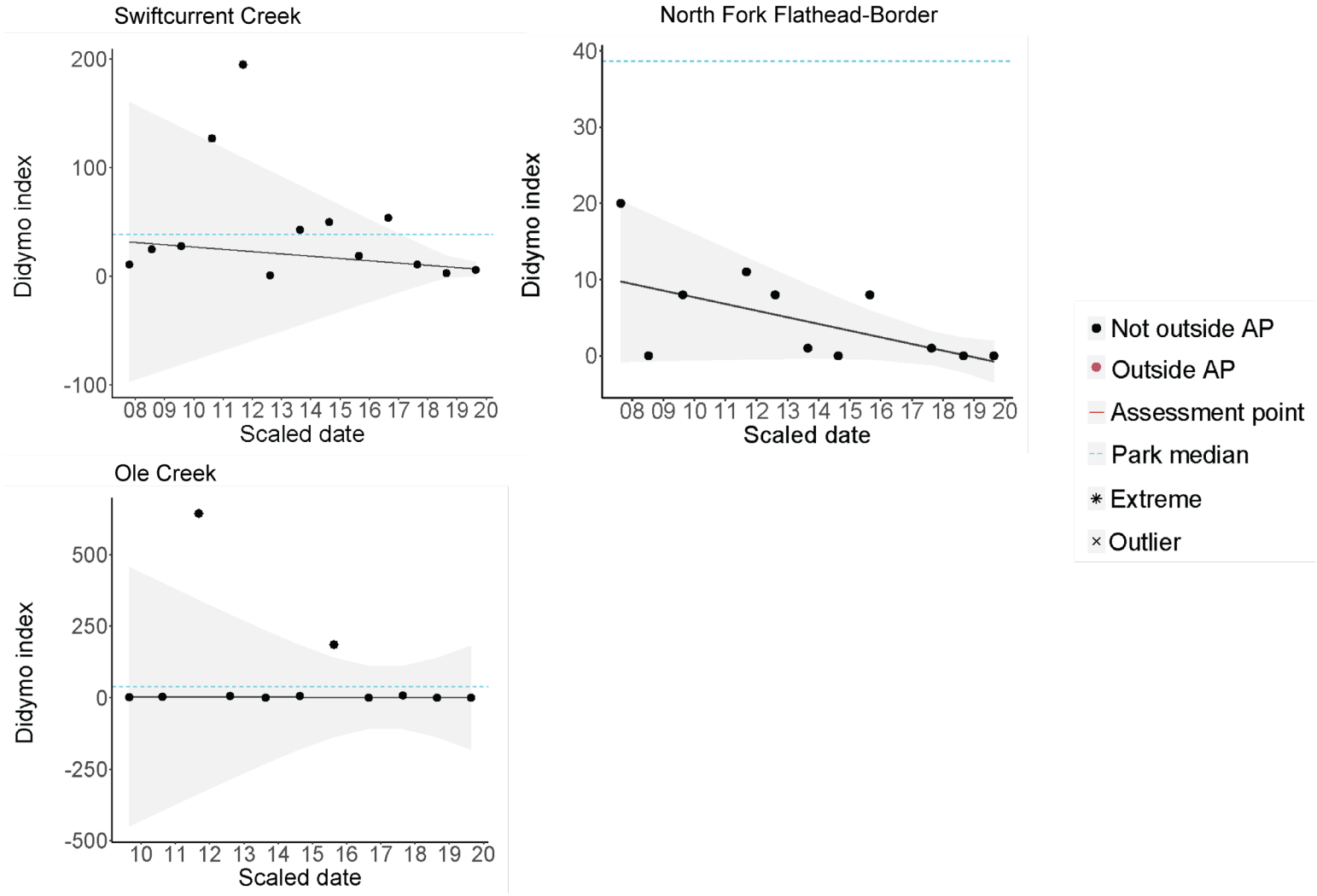


Figure 73. Didymo index in Glacier National Park from 2007 to 2019. (Top left) Swiftcurrent Creek, (top right) North Fork Flathead—Border, and (bottom left) Ole Creek. We omit the assessment point from the plots given its very high value (893) relative to observed data at sentinel site. As available, a dotted blue line shows the park-scale 2007–2009 median. The fitted lines (no significant trends at these sites) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

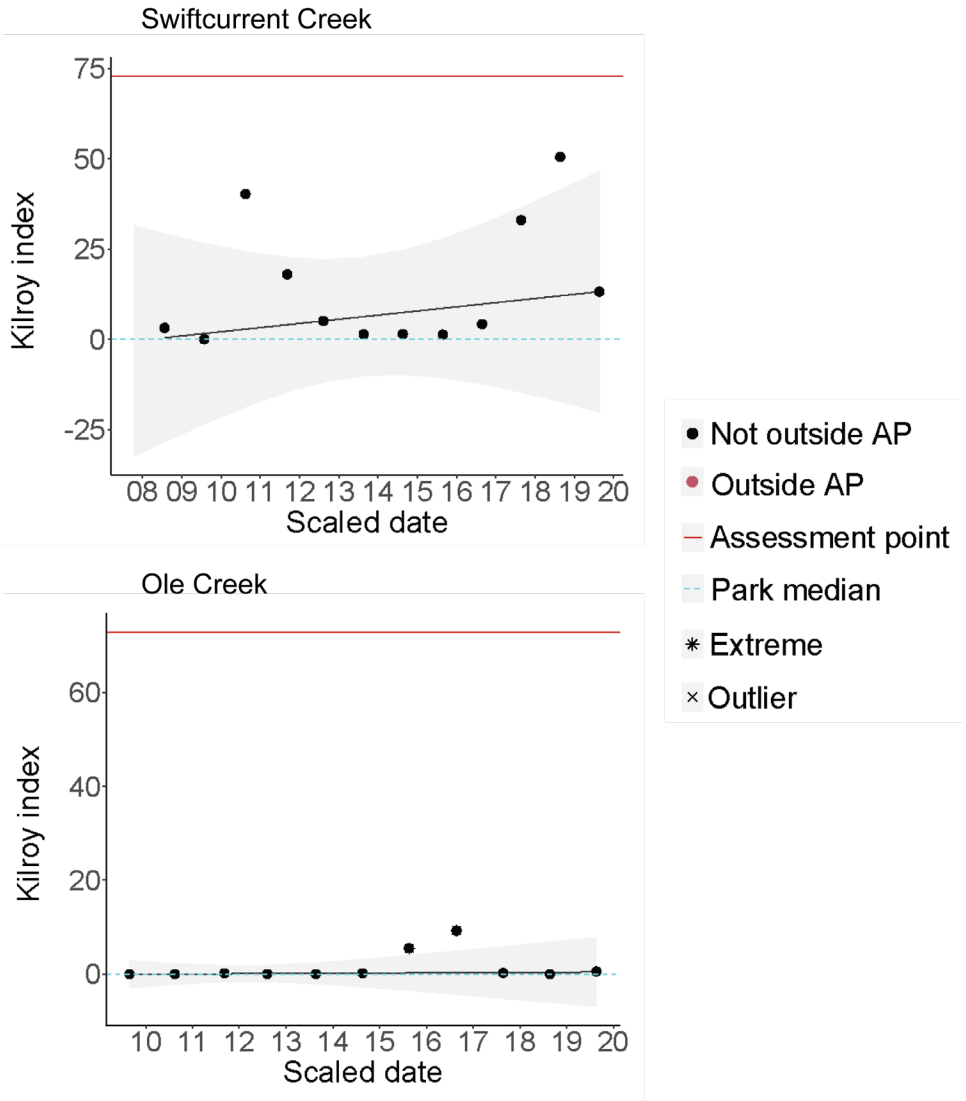
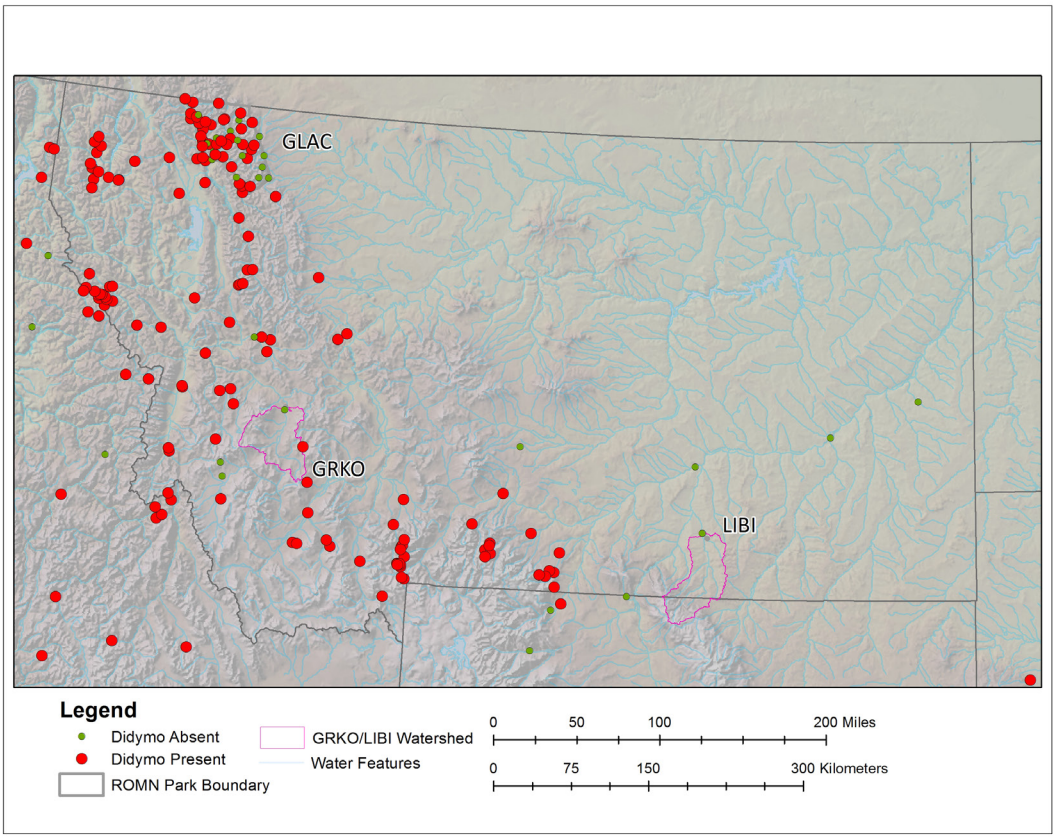


Figure 74. Kilroy index in Glacier National Park from 2007 to 2019. (Top) Swiftcurrent Creek, and (bottom) Ole Creek. As available, red horizontal lines show a non-reference assessment point and a dotted blue line shows the park-scale 2007–2009 median. The fitted lines (no significant trends at these sites) and their pointwise 95% confidence intervals (light gray band) are from a quantile regression model at the median. Not all symbols in the legend are in all plots.

Changes in the regional distribution of didymo are difficult to quantify given lack of a consistent and appropriate sample design across the time and the state, or other larger scales. General data available from the State of Montana and the Montana Natural Heritage Program (State of Montana 2022a) suggest the overall range of didymo has not changed much over the last decade. Confirmed locations have likely expanded in the northwest corner of the state (to the west of Glacier National Park) and, in general, there appears to be a higher density of observations in the mountain corridors of the state (Figure 75).



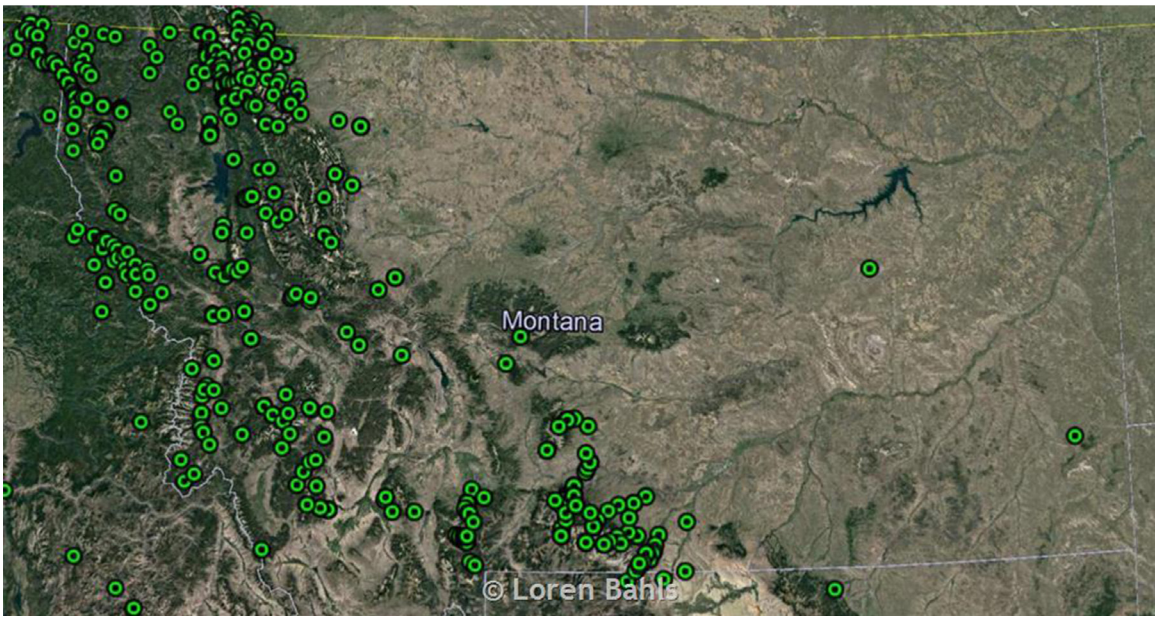


Figure 75. Qualitative comparison of *Didymosphenia geminata* range circa 2009 and 2021. The map in the top panel is from Schweiger et al. (2018) and was compiled from various sources and data spanning 2004 to 2009. The map in the bottom panel is from the State of Montana Field Guide and is likely current through 2021 (State of Montana 2022a).

Condition

No didymo or Kilroy index result was above the reference range suggested by the assessment point (Table 29). Only three individual didymo index values were within an order of magnitude of the assessment point, once at Ole Creek in 2011, and twice at Swiftcurrent Creek in 2010 and 2011. Likewise, only the Kilroy index at Swiftcurrent Creek had more than one meaningfully close value to the assessment point, approaching the threshold in 2010, 2017 and 2018 (Figure 74).

Sentinel sites added in 2018 on the North Fork Flathead—Glacier Rim and on the Middle Fork Flathead do not have sufficient data to estimate trend. As we accrue more data at these sites, we will estimate the same quantile regression models as presented in this report for other sentinel sites. Median didymo and Kilroy index values at these sites were like other sites. However, in 2007 and 2008, didymo index values at the North Fork Flathead—Glacier Rim were relatively high (yet still far below the assessment point). In 2018 and 2019, index values were again quite low. We have never observed a bloom at either of these big river sites.

Discussion and management applications

The didymo story is a complex one. This species has created a bit of an uproar in the scientific literature over the last decade or so and perhaps some heartache (or headache) in natural resource managers in NPS and other land management agencies. It is one of the few freshwater algae species that has grabbed the public's attention, perhaps because didymo blooms can be so visually impressive, often in streams and rivers that otherwise are at least aesthetically, if not ecologically, in a reference condition. It is also interesting from a scientific perspective—there are not many native species (as didymo is in Glacier National Park) that behave in qualitatively invasive ways. This unusual behavior has been described as one of the links to climate change in didymo's ecology.

The timing of the 2007–2009 survey may have hit a high point in didymo distribution and local abundance in the park. These were hot and relatively dry years, with consistently low base flow. Coupled with the park's naturally oligotrophic waters and larger substrates, both preferred by didymo, the conditions for didymo blooms were ripe (Spaulding and Elwell 2007; Schweiger et al. 2011; Jackson et al. 2016). Our large sample of survey sites also included several stream sites near the outlet of lakes—lake influenced hydrology further reduces the amplitude of a site's hydrograph. Sites that instead scour each spring generally remove didymo blooms.

Given these baseline conditions, our ecoregional assessment point, derived using only park reference sites, may not be usefully representative for the park. Additional research is needed that instead links didymo abundance or bloom extent to important ecological responses. The Rocky Mountain Network's ongoing work focused on the macroinvertebrate and diatom MMIs might provide this. Threshold effects between didymo and an MMI could make ideal assessment points, and these would be scaled specifically to Glacier's streams.

Five of our six sentinel sites were sampled as part of the baseline survey. Only Swiftcurrent Creek, located near the outlet of a small lake, and with rapidly warming water, had much didymo in the baseline period. Overall, this generally low presence continued through 2019 across all sentinel sites. However, it may be premature to say that didymo no longer is an issue in the park based on our

sentinel sites alone. Really, we need a resample of the full array of survey and gradient sites to know what didymo has done over the last decade in the park.

In summary, it is difficult to reach any firm conclusions regarding didymo in Glacier—an additional 10 years of sentinel site data has not given us any sense of a final answer. Sentinel sites were generally not part of the parkwide high levels in 2007–2009, and they have, by and large, remained that way. Yet given now documented impacts of rock snot on stream condition (i.e., Ladrera et al. 2018; Jellyman and Harding 2016; Larned and Kilroy 2014), which was not previously well understood, we still feel that this is an issue of importance to the park.

3.5.6.2 Zebra/quagga mussels and New Zealand mudsnails

Zebra and quagga mussels (*Dreissena* spp.) are small shellfish native to Eastern Europe. Zebra mussels were first discovered in North America in the late 1980s in the Great Lakes. Zebra mussels represent one of the most important biological invasions into North America (Benson et al. 2017b). By 2010, they had spread across the Northeast and Southwest United States. Among their many negative impacts, zebra mussels cause serious biofouling by colonizing water supply pipes of hydroelectric and nuclear power plants, public water supply plants, and industrial facilities. Zebra mussels can have profound effects on ecosystems. These were first reported as detections of larvae in Montana in 2016 from Tiber and Canyon Ferry Reservoirs about 100 miles to the east of Glacier.

Large populations of zebra mussels in the Great Lakes reduced the biomass of phytoplankton significantly following invasion. Diatom abundance declined 82%–91% and transparency, as measured by Secchi depth, increased by 100% during the first years of the invasion in Lake Erie (Holland 1993). Biomagnification of Polychlorinated Biphenyls (PCBs) was associated with zebra mussels, indicating that concentration of pollutants in zebra mussel feces can transfer to other trophic levels (Bruner et al. 1994). Zooplankton communities have also been affected by invasions of zebra mussels (MacIsaac and Rocha 1995).

New Zealand mudsnails (*Potamopyrgus antipodarum*) are native to New Zealand. They were first discovered in Idaho in 1987. By 1996, they had reached Yellowstone National Park. They are widespread in California, the lower Columbia River, Washington, in the Colorado River basin, including Boulder Creek near Rocky Mountain National Park, and in central Montana north of Helena. Mudsnails tolerate siltation, thrive in disturbed watersheds, and benefit from high nutrient flows. They can reproduce quickly, mass in high densities, impact trophic dynamics of native trout populations, and alter the physical characteristics of the streams themselves (Benson et al. 2017a).

Assessment points

Given a lack of relevant data for these four species' expected occurrence in Glacier National Park, we do not establish assessment points. However, we recommend the park evaluate whether the assessment point of zero used for invasive plants is applicable to these AIS. Any stream sites with observations of any of these species in the park would be assessed as in a non-reference and, for at least the mussels, emergency, condition.

Parkwide 2007–2009 baseline results

We found no zebra or quagga mussels in our 2007–2009 survey. Likewise, we found no New Zealand mud snails in our 2007–2009 survey.

Trend

We have not found any of these species at any sentinel site over our period of record from 2007 to 2019.

Condition

With no detections of these species in the park at our sentinel sites from 2007 to 2019, this indicator is in reference condition, but given the threat, continued monitoring is needed.

Discussion and management applications

Thankfully, at least at sentinel sites, Glacier National Park continues to be free of these AIS. Regionally, the State of Montana delisted Canyon Ferry Reservoir in 2020 as a suspect water body (State of Montana 2022c, d). Tiber Reservoir has continued to be listed and has ongoing boating restrictions. State and federal agencies, including Glacier National Park, continue to monitor for invasive mussels in hundreds of locations across Montana. There have been no detections as of fall 2019 (Schmidt and McLane 2019). Since the peak New Zealand mudsnail densities in Montana were reported in the early 2000s (State of Montana 2022b), Missouri and Madison River populations have declined over the last decade, but may be building their populations back, especially below Holter and Hebgen Reservoirs (McGuire 2016).

4 Future Directions

4.1 Uses of Rocky Mountain Network stream data

A fundamental goal of Rocky Mountain Network inventory and long-term monitoring is to provide park managers with data and information useful for protecting and managing park resources. Our estimates of trend in stream condition at sentinel sites can help in the understanding of how and what changes have occurred over the last decade and how this might translate to future conditions. This information can help park management report on and meet regulatory requirements, as well as assist with management in the park. Our data can be used to quantify the importance of various "stressors" or "drivers" of stream condition and their trends over time (e.g., nutrient input, sediment loads, climate change-driven changes in stream hydrology, etc.). Specific indicators, when interpreted in the context of the larger stream ecological integrity (SEI) dataset, can also enhance general understanding of stream ecology. Finally, SEI results should also help park staff understand and communicate to the public about the condition and trend over time of stream resources in Glacier. They allow interpretation of specific streams in the context of the broader resource.

Perhaps most importantly, network stream monitoring provides a critical ecological baseline for future assessments. This is enhanced by the comprehensive suite of indicators included in the protocol. As just one example, the park's namesake glaciers—such an important part of stream ecology—are projected to be gone soon. This will change stream temperatures, hydrologic regimes, and sediment loads. SEI monitoring may someday document reduced stream flow, channels with less habitat, altered sediments, and warmer water temperatures. All these changes will likely have detrimental impacts to aquatic communities. We hope network data will offer the park useful guidance on how to interpret and manage these systems in the face of these changes.

4.1.1 Specific recommendations

4.1.1.1 Water chemistry

While most chemistry responses at sentinel sites are in a reference condition, elevated water temperatures and altered hydrologic processes may be impacting stream chemistry. For example, metals in stream sediments were at unexpected levels in our alpine sentinel site, Reynolds Creek, perhaps driven by complex climate change processes. There are also some indications that atmospheric deposition of nitrates is interacting with climate driven changes, impacting pH and alkalinity, and that visitor services may have some possible effects at specific sites by elevating some major ions. *We need additional data, analysis, and interpretation to confirm and support these patterns. We recommend the park partner with the Inventory and Monitoring Division and/or external collaborators to conduct focused research on a possible climate change driven acidity and metals issue at the Reynolds Creek site, as what may be happening in this stream could be happening at a landscape scale in the park and therefore merits additional investigation. In addition, more focused research on possible visitor service driven impacts on water quality at Swiftcurrent Creek may be warranted.*

4.1.1.2 Habitat

Most of our habitat indicators suggest sentinel stream site habitat is intact and in a reference condition. There may be some fine sediment issues at some locations and, using a conservative assessment point, the frequency of invasive/exotic plants was in a non-reference condition at all sentinel sites. Perhaps the most important missing piece in the habitat elements of the SEI protocol is a lack of management relevant and park specific assessment points. *We recommend the park partner with the ROMN and external collaborators to conduct focused research on deriving or better understanding existing assessment points for habitat features like invasive plants, fine sediment, and wood density. In general, any improvement in the connection between park resource management and the assessment points used to evaluate data would enhance the relevance of long-term monitoring.*

4.1.1.3 Bioassessment

Overall, our bioindicators showed that sentinel sites were in largely reference conditions. In some cases, a site's biology allowed us to reconsider and adjust assessments based on chemistry or habitat data alone. We generally consider biological responses as more important—if a stream had intact biology, it likely suggests that most physical or chemical stressors may not really be impacting the site. Nevertheless, bioassessment also revealed results of concern to the park, specifically a negative biological response to warming water, with ecological integrity higher in colder water. *We recommend the park partner with IMD and external collaborators to conduct focused research on temperature mediated impacts to biological communities in the park. Specifically, while we did not see the feared warm water driven "didymo take over" of sentinel sites in the park, we do not have the complete story on this native but invasive diatom. A focused research effort on didymo would give us a better understanding of its potential impacts to the park.*

4.2 Future stream monitoring in Glacier National Park

This effort has been, and will continue to be, a cooperative undertaking between the network and our partners—most importantly, Glacier National Park. We believe our results, representing the efforts of dedicated scientists and resource managers over several years, are valid, representative, and address our objectives. However, our data are still "short," and we are just in the beginning of this effort, even a decade or more in. It will take many more years to have sufficient data to understand long-term trends. As we accrue data, we will also continue to improve our understanding of the mechanisms and drivers behind SEI responses. Future SEI monitoring in Glacier will permit continued analysis of SEI data relative to assessment points from the state and those we developed with park resource management. These analyses will help scientists continue to monitor the most cost-effective and management-relevant suite of indicators of ecological integrity of the streams and rivers of Glacier National Park.

4.3 Glacier as a reference for other landscapes and sentinel of climate change

We conclude this report by highlighting two recurring subjects in our analyses: the role of the park as a reference system, and the role of climate change.

Monitoring complex ecological systems like streams and rivers is always a challenge. This is even more so in landscapes like Glacier National Park that are generally intact, with few classic human

disturbances that lead to clear and unambiguous degraded streams. Assessment methodology in many ways depends on these structural relationships between stress and response to classify the state of streams in the park and their direction of change. However, a clear role for the park and our monitoring data might be to serve as a reference for other streams embedded in more anthropogenic landscapes. Our results can help managers of streams and rivers not lucky enough to be in Glacier aspire to maintain or be restored to a similar condition.

Recent work (Vaughan and Gotelli 2019) presents research that suggests impact from climate change on biological condition may be offset if other stressors on a system are reduced. This is, in a sense, paying climatic debt with accrued environmental credit. Systems like Glacier National Park are of course well positioned to do this—the park already has a rich bank account full of environmental credit given its wilderness context and its largely reference condition. This offers a path forward for park resource management and recognizes that the park's waters remain largely in a pristine state. Several actions are possible, as noted throughout this report, that might enhance the park's ability to mitigate adverse impacts from climate change and improve resiliency. These include continuing to minimize local human use impacts, such as from roads and visitor services. Perhaps most importantly, the park can also help advance public education about the value of intact functioning stream ecosystems, with almost all of Glacier's streams and rivers as shining examples (Figure 76).



Figure 76. NPS Inventory and Monitoring Division and Glacier National Park staff collecting data from Reynolds Creek, Glacier National Park, 2022. (NPS / BILLY SCHWEIGER)

5 Glossary

Algae: Aquatic organisms that photosynthesize but lack a vascular system. Some are microscopic, others are very large.

Aquatic macroinvertebrate or benthic macroinvertebrate: An organism found in waterbodies, frequently associated with stream bottoms, that does not have a spinal column and is visible with the naked eye.

Ash free dry mass: The weight (mass) of a material obtained by first drying the material at 105 ºC and then heating the material to 500 ºC for 1 hour.

Assessment point: A point along a continuum of indicator values where scientists and managers have together agreed to assess or document the status or trend of a resource relative to program goals, natural variation, or potential concerns. The direction of a condition or disturbance state is indicated by an inequality associated with the assessment point. For example a "> 2.0" for a reference state means that values above 2.0 are in reference.

Base flow: The portion of stream flow that comes from groundwater and not runoff.

Baseline: A value or range of values calculated from a time series of data (from one to many years; if many, the time period is often referred to as a "Period of Record"). Baselines may not necessarily be ecological or management assessment points but could be used in a similar way to assess monitoring data.

Beneficial use: A valuable characteristic of a stream or river resource that, directly or indirectly, contributes to human welfare.

Benthic: On or associated with the sediments or bottom of a body of water.

Bioassessment: Using biological assemblage indicators to assess the health or ecological integrity of an ecosystem.

Biomass: The total mass or amount of living or dead organisms in a particular area or volume.

Chlorophyll-a: The major green pigment found in the chloroplasts of plants and algae.

Criteria: In the context of stream ecological integrity monitoring, an assessment point set by well-established regulatory process; may or may not be ecologically or management relevant for a park resource.

Diatom: Any one of a number of microscopic algae, which can live as single cells or in colonies, that are enclosed within two box-like parts or valves (called frustules) made of silica that fit together like the halves of a Petri dish. When diatoms are collected from the bottom substrate of streams, they may be referred to as **periphyton.**

Diel: Involving a 24-hour period that usually includes a day and the adjoining night.

Disturbance class: In the context of stream ecological integrity monitoring, disturbance classes are as follows:

> **Negligible:** A negligible impact is barely detectable and/or would result in no measurable or perceptible changes to streams. Negligible impact corresponds to a minimally disturbed condition (MDC) as used in the ecological monitoring literature.
>
> **Minor:** A minor impact is where an impact is slight but detectable and/or would result in small but measurable changes in streams and/or stream hydrology and the effects would be localized. Minor impact corresponds to a least disturbed condition (LDC) as used in the ecological monitoring literature.
>
> **Moderate:** A moderate impact is readily apparent and/or would result in easily detectable changes to streams and/or stream hydrology and the effects would be localized.
>
> **Major:** A major impact is severely adverse and would result in appreciable changes to streams and/or stream hydrology, with the effects often regionally important.

Driver: Environmental factor that influences (or causes a response in) an assemblage or ecosystem. A driver is a natural factor (as opposed to a human or anthropogenic factor) that is within the "natural range of variability."

Ecological integrity: Ecological integrity is the capacity to support and maintain a balanced, integrated, and adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats of the region.

Ecological threshold: In general, this is a break point(s) or a "break range" that describes a shift in an ecological response measure and bounds regions in a response's distribution. Thresholds often describe meaningful changes in the ecology of a system. They may involve a nonlinear, rapid, or large response relative to the inputs to a system. They often bound a new ecological state from which it is difficult to recover.

Ephemeral stream: A stream or stream segment that flows only in direct response to precipitation in the immediate watershed or in response to the melting of a cover of snow and ice and whose channel bottom is always above the local water table.

Eutrophication: The process of enrichment of a water body by nutrients, usually nitrogen- and phosphorus-containing compounds, and the resulting increase in primary productivity (algal and plant growth and decay). Some definitions include organic enrichment of a water body as part of eutrophication.

Gradient site: In the context of stream ecological integrity monitoring, a stream site sampled once or multiple times to characterize stream condition under a gradient of (primarily) human disturbance—either under stress or in an unstressed pristine condition.

HDI*:* Human Disturbance Index—an index of various field and remotely sensed indicators of human caused disturbance, such as agricultural use, roads, or trails.

In-situ water chemistry: Water chemistry indicator, such as electric conductivity or pH, measured in the field.

Intermediate condition*:* In the context of stream ecological integrity monitoring, an ecological condition between reference and non-reference condition.

Intermittent stream: A stream or stream segment that is below the local water table for at least some part of the year and obtains its flow from both surface runoff and groundwater discharge.

Least Disturbed Condition (**LDC)**: Least-disturbed condition is defined as the best available state within a defined region. In some (if not many) cases, the lack of availability of a minimally disturbed condition (MDC) or a completely undisturbed reference site makes it necessary to resort to the concept of "least-disturbed condition." The LDC may include measurable anthropogenic stress and disturbance. The concept of an LDC implies that there may be an opportunity for management or restoration of sites to an MDC or best attainable state. This may also be possible with an MDC, but in many cases, the set of actions (i.e., reverse climate change) are beyond the scope of management.

Macroinvertebrate: Aquatic invertebrate fauna that can be captured by a 500 µm net or sieve. This includes arthropods (insects, mites, scuds, and crayfish), molluscs (snails, limpets, mussels, and clams), annelids (segmented worms), nematodes (roundworms), and platyhelminthes (flatworms).

Mainstem: The principal river within a given drainage basin where a number of tributaries discharge into a larger watercourse.

Management-based assessment point: In the context of stream ecological integrity monitoring, an assessment point with explicit management applications or where management-relevant changes in a response occurs. Management assessment points may be the same as ecological assessment points, especially in wilderness parks or when there is little or no information about the management relevance of a response (in these cases, we generally refer to an ecological assessment point). They may be set at some estimated distance from an ecological assessment point to serve as a warning ("surveillance assessment point") or to spur management action ("action assessment point").

Metric: Derived forms of an SEI result. For example a characteristic such as the percentage of an assemblage made up by native species of a biological assemblage (e.g., fishes, algae) that changes in some predictable way with increased human influence.

Minimally Disturbed Condition (**MDC)**: A reference condition characterized by the absence of significant human disturbance. In practice, even in wilderness parks, there is usually some background human impact due to diffuse or indirect factors like climate change, nutrient deposition, or residual historical alterations.

Multimetric Index (**MMI)***:* An index constructed of multiple indicators selected in a statistical modeling exercise designed to include only the most relevant, parsimonious set of contributing indicators to a response variable.

Non-reference condition*:* In the context of stream ecological integrity monitoring, a "bad" condition: not intact (e.g., missing some ecological component(s)), unhealthy, having low ecological integrity (used in bioassessment or, in rare cases, where there is a clear functional ecological understanding of the relationship between an indicator and a stressor or impact). Primarily influenced by anthropogenic drivers.

Perennial stream: A stream or stream segment that has flowing water year-round, except during extreme drought.

Periphyton: The microscopic flora and fauna that grow at or are associated with the bottom of a body of water and include microscopic algae, bacteria, and fungi.

Phytoplankton: Free living, generally microscopic algae commonly found floating or drifting in waterbodies such as the ocean, lakes, and streams.

Point source: A discernable, confined, and discrete conveyance, including but not limited to any pipe, ditch, channel, tunnel, conduit, well, discrete fissure, container, rolling stock, or vessel or other floating craft, from which pollutants are or may be discharged.

Reference condition*:* In the context of stream ecological integrity monitoring, a "good" condition: intact, healthy, having high ecological integrity (used in bioassessment or, in rare cases, where there is a clear functional ecological understanding of the relationship between an indicator and a stressor or impact). Primarily influenced by natural drivers. The region of a response distribution on the "good side" of an assessment point (usually ecological). The reference condition is a distribution as it encompasses natural variation in the response. It is somewhat synonymous with a "natural range of variability." Some authors restrict reference condition to describe a biological response, but the network proposes using it more loosely to refer to any type of response measure. Reference conditions are defined relative to a specific response, target population, and spatiotemporal scale (i.e., this is different for a stream response in Glacier National Park than for the Northern/Canadian Rockies Ecoregion). Based on the dominant landscape condition, reference conditions may reflect various degrees of background disturbance.

Response measures: Stream (or other) indicators that "respond" to natural (e.g., drought) and human (e.g., agricultural development) drivers and stressors. In contrast to a metric, a response measure is not the result of a calculation.

Salmonids: Ray-finned fish, whose members include salmon, trout, chars, freshwater whitefishes, and graylings.

Saturation: A state in which the gas concentration (e.g., oxygen) in a water body is in equilibrium with the local partial pressure of that gas in the atmosphere.

Sentinel site: In the context of stream ecological integrity monitoring, a targeted or hand-picked site sampled (usually annually) to help understand stream dynamics or important for management purposes; results cannot be inferred to other streams.

Standards (water quality standards): In a water quality regulatory context, a term applicable to state waters, referring collectively to their designated beneficial uses, criteria, and the non-degradation policy, all in Montana law. **Numeric water quality standards** are quantified expressions of water quality in state law intended to protect a designated beneficial use or uses.

Status: In the context of stream ecological integrity monitoring, status is a "snap shot" evaluation of an indicator over a well-defined and concise range in time.

Stressor: Environmental factor that influences (or causes a response in) an assemblage or ecosystem. A stressor is a human influence or a natural factor that, because of human action, is outside the "natural range of variability."

Survey: Random probabilistic sampling of a resource (i.e., streams) at intervals (e.g., every 10 years) such that results provide inference to the entire resource; applies to "survey sites" sampled in this analysis.

Trend: A non-cyclic, directional change in a response measure that can be with or without pattern.

Wadeable: A stream whose Strahler order is first through (at most) sixth (1:100,000 map scale), and in which most of the wetted channel is wadeable by a person during base flow conditions.

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Appendix A: Assessment Details

Assessment is the process by which the condition of an ecological resource, such as a single stream or a population of streams in a park, is placed into classes that describe or interpret condition. We assess the condition of Glacier National Park's streams relative to "assessment points." Assessment points (APs; Bennetts et al. 2007) are numeric values of an indicator where scientists and park managers have agreed to evaluate the status or trend in condition of a vital sign. Assessment points can be defined or set using several methods, including (in general order of preference): 1) direct definition by park management, 2) adoption from regulatory criteria, or 3) via application of ecological theory (which can overlap with the first two methods). However, none of these forms of AP were readily available for the stream vital signs presented in this report. Therefore, for Glacier's stream indicators, we use two additional methods: 4) development of APs from distributions of indicator data at stream sites that are known or expected to be undisturbed by human activities (these are known as "ecoregional assessment points"), and 5) APs defined by other authors using a variety of methods.

The following summarizes and updates our approach (from previous reports and the Rocky Mountain Network's wetland ecological integrity and the stream ecological integrity protocols) for application in this report. We focus on how ecoregional assessment points are derived and on the miscellaneous methods developed by other authors that we apply to our data.

Ecoregional assessment points

For select Glacier National Park stream indicators, we define ecoregional APs from distributions of data at stream reference sites known or expected to be undisturbed by most human activities from across the Northern and Canadian Rockies Ecoregions in Montana (Figure 77). Ecoregional reference sites include a gradient of important natural disturbances—much if not all the variation in an indicator across reference sites is due to natural sources. The process of identifying reference sites is a key step in the ecoregional assessment point methodology. Stoddard et al. (2006) summarize multiple conceptual considerations and terminology relevant to the process and applied methods are described in various US EPA and Montana DEQ protocols (Stoddard et al. 2005; Suplee et al. 2005). In summary, undisturbed sites are identified via best professional judgement and various statistical models, including relatively simple filtering of sites based on geospatial (i.e., land use) and site level (i.e., soil chemistry) data. Importantly, these filtering tools should not include a known or estimated biological condition of candidate sites to avoid circularity. Reference sites used in this report do include a subset of our long-term monitoring sites in Glacier defined as reference based on methods from Montana DEQ (Suplee et al. 2005) as applied in Schweiger et al. (2018). As with other ecoregional reference sites, this classification is determined independent of biological condition.

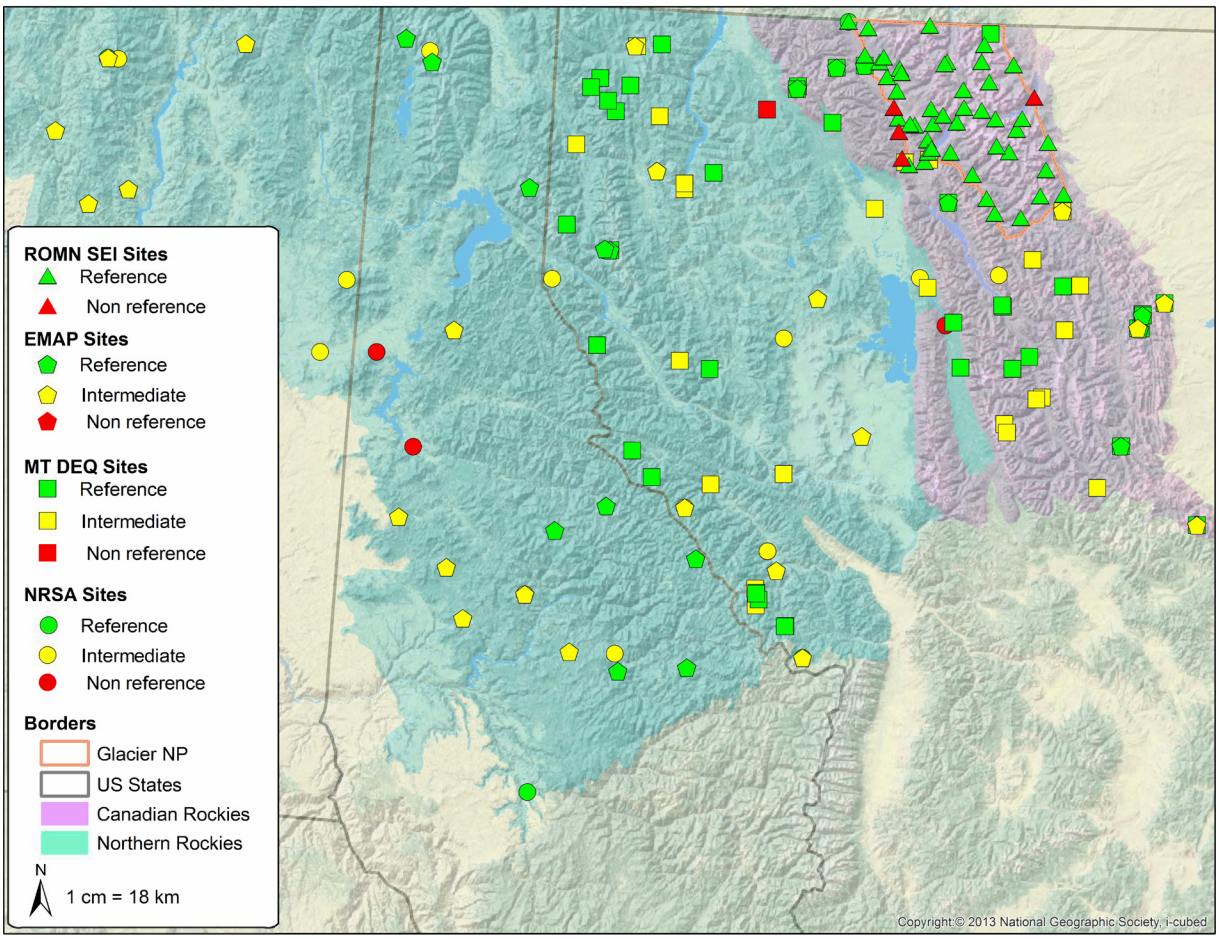


Figure 77. Ecoregional stream and river sites in the Northern and Canadian Rockies Ecoregions used to estimate ecoregional assessment points for stream water chemistry, physical habitat, and biological variables. Not all sites have all data types. ROMN SEI = Rocky Mountain Network stream ecological integrity monitoring sites, EMAP = EPA Environmental Monitoring and Assessment Program, MT DEQ = Montana Department of Environmental Quality, NRSA = EPA National Rivers and Streams Assessment. Most sites were sampled between 2001 and 2009. Some sites were sampled by more than one program. Sites are classified into reference or disturbance classes by the program that established the sites. Sites classified in the reference or minor disturbance category are used to estimate ecoregional assessment points used in this report.

Ecoregional APs are defined in distributions of data at reference sites following US EPA (Stoddard et al. 2005, 2006) via percentile values that bound condition or disturbance classes in a dataset. The most salient conceptual consideration from Stoddard et al. (2006) for the process of identifying undisturbed sites is the degree of ambient human disturbance in and around a park. Some parks are embedded in heavily human used landscapes, while others are largely surrounded by wilderness. The level of ambient human disturbance in and around a park affects the distribution (i.e., shape and location) of data sampled at ecoregional (and park) reference sites, and this, in turn, impacts how well these sites represent a true undisturbed state. We can visualize a hypothetical distribution of data via cumulative distribution functions (CDF; Figure 78). A CDF (loosely speaking) shows data in

increasing order on the x-axis as a percentage of the total on the y-axis. For an indicator collected at sites with more ambient human-caused disturbance (Figure 78, black CDF), and that increases in value with condition (or decreasing human stress), the CDF will likely be expanded or spread out and shifted to the left and further from the true undisturbed distribution (Figure 78, red CDF). Less human disturbance (Figure 78, blue CDF) likely reduces overall variance and moves the distribution to the right or "towards truth." More ambient disturbance also reduces the likelihood that a set of undisturbed sites truly reflects an undisturbed condition (Figure 78—fewer parts of the black CDF than the blue CDF overlap with the red CDF). Stoddard et al. (2006) and the Rocky Mountain Network SEI protocol uses jargon for these two ambient disturbance levels. A context with more human disturbance is labeled a "least disturbed condition (LDC)" (Figure 78, black points), while less disturbance is called "a minimally disturbed condition (MDC)" (, blue points). We include these terms here for consistency but have found that given the similarity or connotations of the words "least" and "minimal" they are not so useful and we do not emphasize them beyond how they help structure the ecoregional assessment process we use.

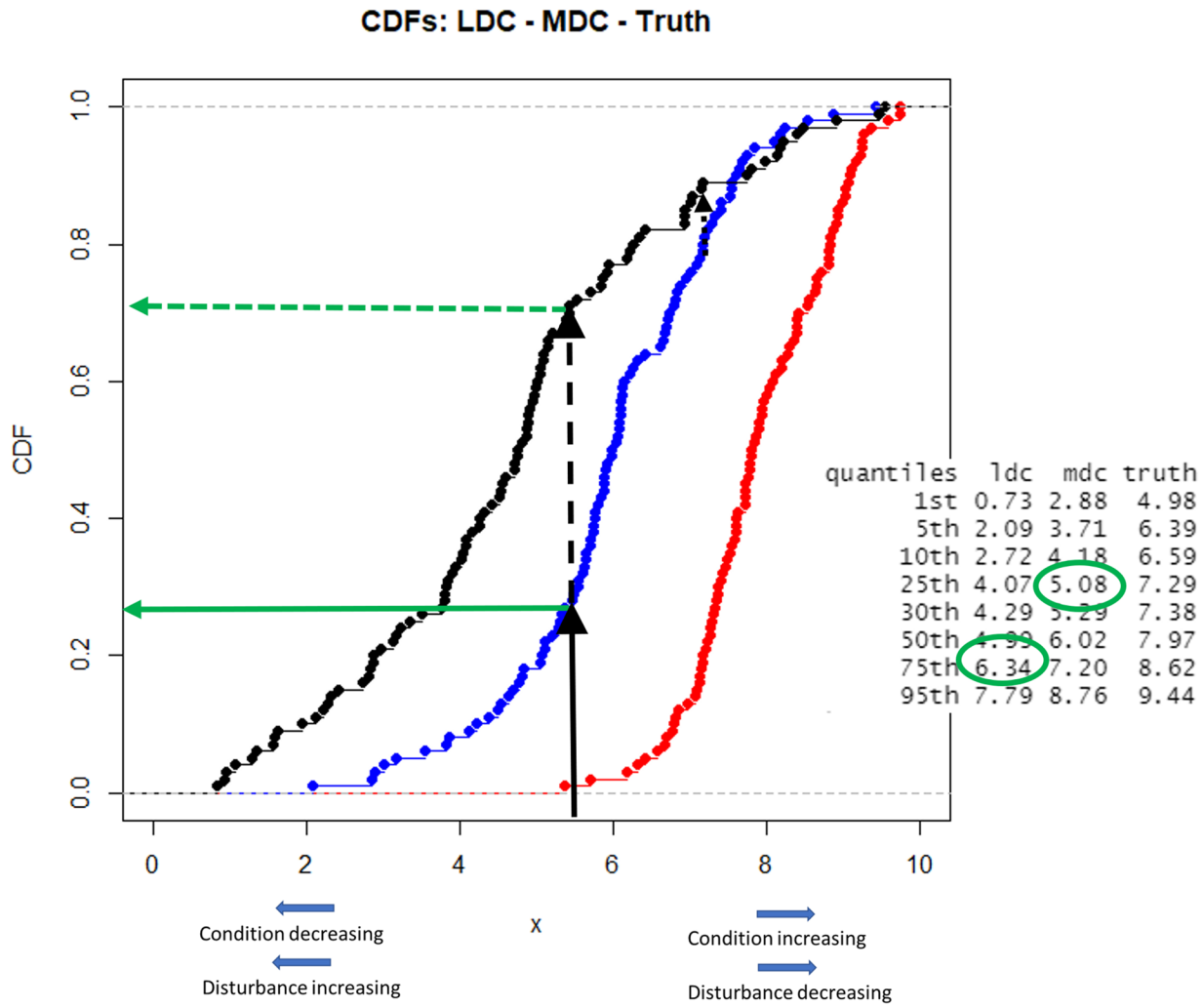


Figure 78. Hypothetical empirical cumulative distribution functions (CDF) simulated from a truncated normal distribution for an example indicator that increases in value as condition increases (or disturbance is reduced). Black points/CDF represent a more human disturbed state or the "least disturbed condition (LDC)," a less disturbance or the "minimally disturbed condition (MDC)" is represented by the blue point/CDF, and a theoretical true undisturbed state is shown in the red points/CDF. The table and vertical black arrows give the lookup value at the 25th percentile for the MDC (5.08). They also show that for the LDC to match this, the percentile used must be increased to around a 75th percentile. This is similar to when the percentile must be increased to elevate the assessment point generated from a more human influenced distribution of undisturbed sites (i.e., LDC).

Lookup percentile values for ecoregional APs are defined or set following US EPA (Stoddard et al. 2005, 2006). For example, for an indicator like most biological or habitat metrics that increase in value with increasing condition (or decreasing disturbance), the 25th percentile is used to define a "minor" disturbance class or reference condition, while below the 5th percentile is defined as a "major" disturbance class or non-reference condition. For indicators that increase with decreasing condition (or increasing disturbance), such a most water quality variables or HDI, the opposite end of

the distribution is used. Values below an AP at the 75th are in "minor" disturbance or reference condition, while values above the 95th means a "major" disturbance or non-reference condition class.

Our ecoregional AP methodology further defines the choice of the percentile lookup values based on whether the undisturbed sites used are taken from a park and/or its ecoregion that are in a highly disturbed (LDC) or less so (MDC) context. Ecoregions (and the parks within them) with less disturbance (MDC) default (following Stoddard et al. 2005) to 5th/25th percentiles as APs for major and minor disturbance classes, respectively. Ecoregions with more disturbance (LDC) use percentiles that are increased above those defined for an MDC. Non-reference moves from the 5th to the 25th, while a reference condition AP is increased from above the 25th to the 50th. Increasing the percentiles moves the resulting AP closer to the true undisturbed state (i.e., Figure 78) because a more disturbed distribution is likely further away from an undisturbed state and thus more appropriately scores disturbance in landscapes with more disturbance. Stoddard et al. (2005) illustrate this effect using a large sample of undisturbed sites in two different contexts—less disturbed mountain ecoregions and more disturbed plains ecoregions. In the plains, APs are based on higher percentiles, and this moves the boundaries of the condition classes closer to what they likely are in a true undisturbed state. Importantly, all of this is specific to a given indicator—some variables will be less affected by disturbance and some more within a single ecoregion. Thus, a park may use AP percentiles based on less disturbance for one indicator and more for another.

The choice of whether a given indicator in Glacier National Park is in an MDC or LDC context (setting the AP percentile look up values) is an important step in the assessment process and should ultimately be made by a park in consultation with the Rocky Mountain Network. Schweiger et al. (2018), in consultation with the park, set this as an MDC for every indicator we analyzed, and, with select exceptions as noted in each section below, we continue this approach in this report. For example, for an indicator like a biological metric that increases in value with increasing condition (or decreasing disturbance), the 25th percentile is used to define a reference or minor disturbance class, while a non-reference or major disturbance class is set below the 5th percentile. Values that fall between these two thresholds are considered intermediate in disturbance.

We interpret comparisons between an indicator and its AP in a collaborative way with park staff and management in the context of park management objectives, NPS Inventory and Monitoring Division guidance, NPS resource management policies, and general ecological theory. With the exception of the Montana DEQ integrated nutrient assessment, we do not perform statistical tests of SEI data or trend estimates relative to an assessment point. Comparisons are instead whether an SEI response or some proportion of responses over time, is above or below an assessment point, with no estimate of the statistical probability of this occurrence.

Appendix B: Analytical Method Details

Water balance

We estimate the components of the water cycle (Figure 79) using equations outlined by Lutz et al. (2010) as implemented in Thoma et al. (2020). We first derive precipitation and temperature from DAYMET version 4 (Thornton et al. 2020) at 1 km^2 index grid cells placed near the top of each site's catchment. The water balance model partitions precipitation (P) falling on this cell into rain or snow following Jennings et al. (2018), the latter of which accumulates until temperatures become warm enough to melt (Hock 2003). Subsequent steps in the water balance process treat the movement of water from this grid cell into the catchment and a stream sentinel site.

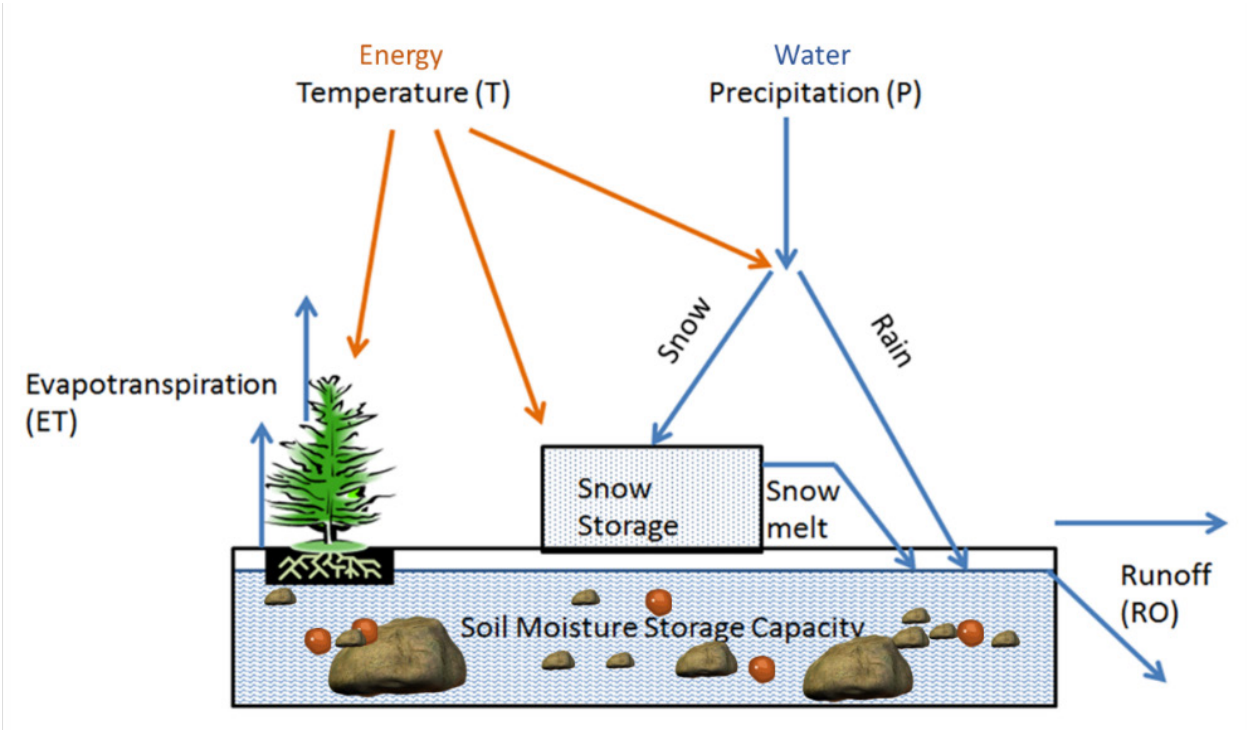


Figure 79. Water balance schematic adapted from Gray and McCabe (2010). Direct runoff—precipitation that leaves the system without entering the water cycle—and climatic deficit are not shown. Soil moisture estimated by the water balance model and water holding capacity take from SSURGO data are integrated as Soil Moisture Storage Capacity.

We estimate an initial direct runoff parameter, or the portion of any precipitation that does not enter the water cycle but instead leaves the system, via methods developed in NRCS (2017). The water balance model then calculates a potential evapotranspiration via the Oudin method as the amount of water that could be evaporated and transpired with available energy if water was unlimited (Oudin et al. 2010). Potential evapotranspiration uses slope and aspect summarized across the catchment as derived from a 10 m digital elevation model to better estimate the localized heat load used in deriving potential evapotranspiration. We next estimate an initial soil water holding capacity at stream sites as the quantity of water stored in the top meter of soil as defined by SSURGO soil data (NRCS 2021).

Iteration of the model acts upon this soil water via actual evapotranspiration, weighted by an estimate of shade cover from vegetation derived from riparian cover estimates from SEI monitoring data. A climatic water deficit is estimated as the amount of additional water vegetation would use if available, calculated as the difference between actual and potential evapotranspiration (Stephenson 1998). The water balance model then estimates water that exceeds soil storage capacity as the runoff within the water cycle (e.g., not leaving the water cycle, as direct runoff does). Finally, runoff values are used to accumulate a "drainage" term, following Croke et al. (2005). Drainage is built as a unit hydrograph with coefficients for "quick" or event-based flow, and "slow" or base flow.

Several parameters in the water balance model come from readily available GIS data, such as slope or aspect, or from field data, such as vegetative cover. These scale the relatively coarse climate (T and P) parameters from DAYMET to more localized edaphic conditions and likely improve the precision of our water balance data. We also conduct a series of model calibrations to improve water balance output relating drainage or other water balance variables to stream flow and/or water temperature. Trial work has suggested that the direct runoff and water holding capacity parameters are most sensitive within this calibration. We optimize bivariate fits of these terms to response(s) by iterating these parameters singly and as pairs across a range of values, maximizing adjusted R^2. We then do the same for the "quick" and "slow" a and b drainage coefficients to calibrate drainage—these coefficients can have a strong effect on how well drainage fits a response. This calibration may be done in a relatively simple heads-up method using initial conditions that we have seen work well for typical Rocky Mountain Network streams and rivers. It may also be done using an optimization routine that steps through all possible combinations of parameters checking for the best relationship between drainage parameters and stream flow and/or water temperature.

Water temperature data development

Table 30 presents a summary of the water temperature data developed for this report. Most time series were from Rocky Mountain Network loggers, although two sites on the North Fork of the Flathead River had meaningful historical data from USGS that we also used. These two sites had longer periods of record extending into the 1980s or 1990s. At the other three sites, the network period of record was around 10 years. At two sites, network and USGS temperature data overlapped and had highly significant relationships.

The raw data for all network loggers was hourly or 30 minutes in frequency. To avoid intractable datasets, we elected to summarize these data to a daily time step. We calculated a difference between day and night temperature using the raw data, but otherwise do not further treat raw data. We then developed test models at daily, weekly, and monthly time steps. All these options resulted in useful results, but we assume that a daily time series is most reflective of the ecology of the site, and daily models were better at describing the total variation in a dataset. Weekly models were a viable option, and as these datasets grow, may be preferable.

Exploratory data analyses (EDA) suggested water temperatures restricted to an ice-free period or approximate growing season were most responsive and revealed trend and relationships with predictors in a statistically better and ecologically more interpretable way. We therefore limited all data to days from late spring or early summer to early fall, depending on the site.

Table 30. Summary of water temperature data source, period of record (POR), and summary statistics by year and approximate ice-free periods as used in models for sentinel stream monitoring sites in Glacier National Park. ROMN = National Park Service, Rocky Mountain Inventory and Monitoring Network. USGS = US Geological Survey.

Site	POR	Approximate Ice-free period	Number of Ice-free records	Data source	Adj R2 of ROMN-USGS overlap	Annual mean (SD)	Ice-free mean (SD)	Annual mean maximum (SD)	Ice-free mean maximum (SD)
Swiftcurrent Creek	5/17/2010 8/26/2019	May 1 to Oct. 31	1681	ROMN - 98% USGS - 2%	0.92	5.03 (5.039)	8.83 (4.319)	6.35 (5.979)	10.89 (4.968)
Reynolds Creek	7/13/2009 8/19/2019	June 1 to Sept. 15	1272	ROMN - 100% USGS - 0%	NA	2.47 (3.231)	5.47 (3.811)	3.77 (5.169)	8.81 (5.969)
Ole Creek	5/17/2010 8/19/2019	May 1 to Oct. 31	1147	ROMN - 100% USGS - 0%	NA	4.28 (3.511)	7.34 (2.364)	6.00 (4.652)	9.95 (3.154)
North Fork Flathead—Border (USGS data) [a]	1/1/1980 [b] 9/29/1991	May 1 to Oct. 31	1991	ROMN - 0% USGS - 100%	No overlap	5.36 (4.641)	8.91 (3.282)	6.58 (5.353)	10.74 (3.790)
North Fork Flathead—Border (ROMN data) [a]	5/12/2010 12/31/2019	May 1 to Oct. 31	1376	ROMN - 100% USGS - 0%	No overlap	5.296 (4.529)	8.73 (3.267)	6.58 (5.353)	10.81 (3.765)
North Fork Flathead—Border (all data) [a]	1/1/1980 [b] 12/31/2019	May 1 to Oct. 31	3367	ROMN - 42% USGS - 58%	No overlap	5.33 (4.594)	8.89 (3.279)	6.60 (5.422)	10.76 (3.778)
North Fork Flathead—Glacier Rim	1/1/1998 [c] 9/30/2019	May 1 to Oct. 31	6320	ROMN - 4% USGS - 96%	0.99	7.72 (5.441)	11.15 (3.778)	8.74 (5.754)	12.32 (3.975)
Middle Fork Flathead [d]	8/27/2018 [e] 8/19/2019	May 1 to Oct. 31	484	ROMN - 100% USGS - 0%	NA	7.26 (2.823)	9.24 (0.937)	7.85 (2.529)	10.14 (1.024)

[a] Site North Fork Flathead—Border is presented in two roughly equal intervals given the long gap in between the 1980 to 1991 USGS data and the 2010 to 2019 Rocky Mountain Network data. We also include an overall set of values for North Fork Flathead—Border.

[b] USGS data at site North Fork Flathead—Border began in in 1975 but we trimmed this to 1980 to match the available record for water balance. No water temperature data were collected at North Fork Flathead—Border from 9/29/1991 to 5/13/2010; North Fork Flathead—Border loggers were stolen in 2017 and then again in 2018 and no data are available for these years. North Fork Flathead—Border installation was moved in 2018 and has not been disturbed since.

[c] We dropped 1996 and 1997 from site North Fork Flathead—Glacier Rim given indications of QAQC issues.

[d] Episodic instantaneous water temperatures from 2001 to 2010 are available from USGS for Middle Fork Flathead site. This smaller sample size did not allow GAM models and should be considered when comparing across sites.

[e] Episodic instantaneous water temperatures from 1982 to 2013 are available from USGS—see main text.

Because thermal responses can lag other environmental phenomenon, we estimated lags between water temperature and a climate, water balance, or stream flow predictor via bivariate linear models with lags of 1 to 365 days. For instance, if water temperature in a stream of a given watershed size is affected two days after a precipitation effect due to delayed runoff dynamics, the flows two days prior to the recorded temperature represent the lag and were incorporated into the model. If a lagged relationship significantly increased the adjusted R^2 more than a contemporaneous version, we included this lag as a candidate predictor. For most models and most predictors, there were lags that fit these criteria. We limited lagged terms to a single instance unless there was a clear indication that two lag structures were present, typically one positive and one negative across two different periods.

Water balance calibration

Creation of the suite of water balance metrics used here included calibration to water temperature and/or stream flow (Table 31). This is possible or even required because several of the parameters in the water balance model are not knowable with a reasonable degree of accuracy (e.g., we do not have field-based measures of soil water holding capacity) and so an iterative process of adjusting parameter values within a reasonable range can improve final metrics. Heads up and automated calibration (where all possible combinations of parameter values can be checked) of direct runoff, water holding capacity, and the a/b coefficients of quick and slow drainage components were evaluated with linear and/or logarithmic fits to water temperature. These bivariate models are not the same as the final GAM models presented below—they do not account for autocorrelation, covariance structure amongst predictors, or possible lags in relationships. We therefore did not expect the adjusted R^2 values to be as high as in the final models where all these complications are statistically included, and we present monthly values in Table 31 as some of these complications are reduced with monthly data (i.e., autocorrelation is reduced). Where we had continuous stream discharge, we also examined relationships of discharge to these parameters in time series plots (see Figure 80 for an example for North Fork Flathead—Glacier Rim site). Adjustments to the a/b coefficients of quick and slow drainage were made to best match patterns in drainage to the limbs of the hydrograph.

Table 31. Summary of water balance calibration at stream ecological integrity sentinel monitoring sites in Glacier National Park, 2007–2019. Latitude-Longitude is for an index site used to generate climate from Daymet, located by hand near the top of each site's watershed. Slope and aspect are summarized from a 10 m National Elevation Dataset cross each site's catchment. Initial water holding capacity (WHC) was estimated following NRCS (2021) using landcover from the National Landcover Database and Hydric Soil Class from SSURGO soil data for the soil polygon at each site. Initial direct runoff (DRO) was estimated from SSURGO Soil Maps. Calibration of WHC and DRO was iterative beginning from the initial values estimated for each. The shade fraction is the inverse of the mean (across all samples in the period of record) total riparian cover from stream ecological integrity plot data. A Jennings (Jenn.) T50 parameter used in estimating effect of rain on snow events is summarized across each site's catchment from a national scale grid developed in Jennings et al. (2018). Quick and slow flow a/b coefficients following Croke et al. (2005) are calibrated by hand or via an optimization routine in R. We set a wind speed parameter to a constant value of 1 for all sites (or no wind effect). We used a value for a Hock coefficient of 4 to help estimate the melting of snow to water for all sites as suggested in Hock (2003). R^2 is for the fit of the final total drainage expressed in cubic feet per second to the maximum monthly ice-free period temperature.

Site name	Index site latitude	Index site longitude	Slope (°)	Aspect (°)	Initial WHC (mm)	Calibrated WHC (mm)	Initial DRO (%)	Calibrated DRO (%)	Calibrated quick flow a/b	Calibrated slow flow a/b	Shade fraction	Jenn. T50	R^2 Drainage to max. monthly summer temp.[a]
North Fork Flathead—Glacier Rim	49.425985	−114.72615	18.25	174.66	65.60	200.00	3.88	3	0.9999/ 0.00001	0.9770/ 0.0207	0.93	1.582	0.54 [b]
North Fork Flathead—Border	49.425985	−114.72615	18.20	170.17	79.30	200.00	4.44	3	0.961/ 0.01	0.97555/ 0.0182	0.98	1.594	0.70 [b]
Swiftcurrent Creek	48.784396	−113.75388	28.40	177.44	89.30	90.00	0.87	1	0.99/ 0.0009	0.98/ 0.0182	0.79	1.796	0.84
Reynolds Creek	48.691718	−113.73494	14.47	109.60	66.40	5.00	4.71	85	0.93/ 0.009	0.9770/ 0.02	0.86	1.582	0.62 [c]
Ole Creek	48.400268	−113.35916	25.60	205.88	89.30	90.00	18.55	30	0.97/ 0.0089	0.9798/ 0.0142	0.7	1.582	0.84

[b] For sites North Fork Flathead—Glacier Rim and North Fork Flathead—Border, we limited the comparison of drainage to maximum summer water monthly temperatures from 2007 to 2019.

[a] Fits are to synoptic discharge measurements summarized to months.

[c] The fit of drainage to maximum summer monthly water temperatures at Reynolds Creek had a large lag that reduced the R^2 value. This was accounted for in the GAM model.

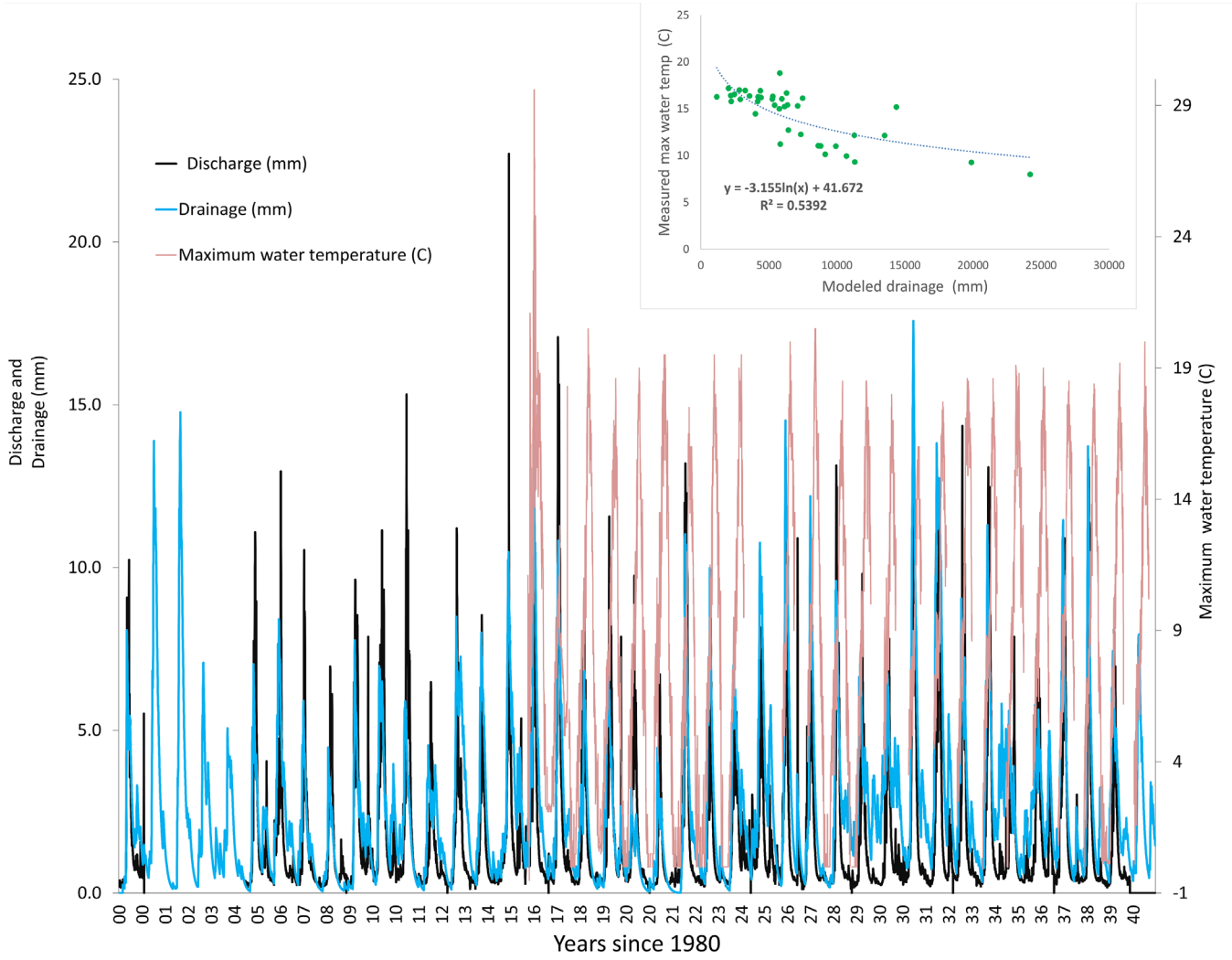


Figure 80. Example calibration of modeled drainage to measured discharge, North Fork Flathead—Glacier Rim site in Glacier National Park, 1980 to 2019. Main plot shows daily discharge flow from the USGS gauge, drainage and maximum daily water temperature time series. Inset shows a bivariate fit between monthly maximum water temperature drainage (logged) over our period of record (2010 to 2019).

Estimating trend

The general form of trend models used to analyze stream ecological integrity (SEI) data depends on several attributes of the data. Foremost among these is the sampling interval or frequency of data collection. Our methods break into two forms: 1) data that are synoptic or sampled once to several times a year, and 2) data that are a time series or can be treated as continuous.

Responses with one to several replicates within a year: quantile regression

Modeling ecological data across time can be a challenge as data often do not meet the assumptions of many statistical tests. Exploratory data analysis (EDA) of chemistry, habitat, and biological response collected once to several times a year suggested that these data had these challenging characteristics. EDA also indicated that quantile regression (QR) would be an ideal model form given the structure of SEI data. Alternate methods such as linear regression have limitations, with data influenced by outliers, and may therefore provide a poor description or prediction of pattern. Other nonparametric methods, such as the seasonal Mann Kendall test, are unable to easily include covariates important in understanding or correcting for issues in data. QR makes no assumptions about the distribution of the underlying data or residuals and is robust to outliers in the dependent variable. QR does assume the dependent variable is continuous. There are many cases, often common in chemistry and other SEI responses, such as skewed data, multimodal data, or data with outliers, when the behavior at the conditional mean fails to fully capture the patterns in the data.

Quantile regression (QR) was introduced by Koenker and Bassett (1978) as an alternative to linear regression and has become an important statistical tool. QR is a semiparametric model form that models conditional quantiles of a dependent variable (linear regression models only the conditional mean). Quantiles are derived from the rank order of values in a distribution with, for example, the middle value of a sorted sample is equal to the 50^{th} percentile (aka the median). In QR the coefficients of each predictor are functions with a dependency on the estimated quantile, and estimating coefficients involves minimizing the median absolute deviation from the modeled quantile. All models presented in this report use a QR at the median. Future work may examine other quantiles, such as the 5^{th} or 95^{th} percentile of a response—these may provide valuable insights where answers to important stream and river resource managements questions in Glacier National Park lie in modeling the tails of a conditional distribution.

Exploratory data analysis

EDA included a variety of exploratory work. We created and visually examined simple time series plots. We generated box plots by lab, year, and season (where applicable). We examined distributions via nonparametric tests of normality. We ran tests of homogeneity across year/lab/season groups. We constructed simple linear models to explore potential patterns in a response and possible covariates. We estimated the relative importance of the terms in each of the EDA linear models and visually examined partial residual plots. We examined the residuals from these linear models for normality and patterns in the relationships between residuals and fitted values. Select EDA results are presented throughout this report; however, the linear models are not presented as they were improved and replaced with the more robust and statistically appropriate QR models.

Quantile regression details

For chemistry responses we initially conducted model selection across a set of nested potential predictors, including stream flow (or water balance derived drainage, hereafter referred to as flow), season, lab, and a lab by time interaction. However, there were often unclear solutions with difficult to interpret combinations of predictors across sites and/or parameters. We therefore reverted to a more consistent structure that always included stream flow, season, lab, and a lab by time interaction as a defined set of terms for all models. We only applied model selection to the choice of including stream flow as a log or as untransformed values. We used the change in Akaike Information Criterion (AIC; Venables and Ripley 2002) with a threshold of 2.0 to choose between a logged or not logged flow term. Where the delta AIC was not above this threshold, we defaulted to the simpler case of not logging flow. Including flow in all models incorporates a useful explanatory variable, generally improves the power of the statistical test, and decreases the possibility that the observed trends are artifacts of variability in streamflow (Hirsch et al. 1991; Schertz et al. 1991). Note there are fewer model assumptions to evaluate for QR models than for a traditional linear regression. We do evaluate autocorrelation in the residuals using autocorrelation (ACF) and partial autocorrelation (pACF) functions.

With our approach to model structure, models could include nonsignificant (i.e., $p > 0.05$ as discussed above) effects of flow, season, lab, or lab by time interactions. This allows statistically nonsignificant but likely ecologically relevant effects to still matter in the estimation of trend, and, perhaps most importantly, control of marginal effects such as lab and/or any interaction between lab and time. A subset of responses did have a unique a priori log transformation of the response variable. Schertz et al. (1991) recommends total P, total N, and TSS be estimated with a logged response given a strong connection to sediment dynamics as influenced by flow. We could not include this as a model selection step as models with different response distributions are not nested as required. For these parameters we estimate models as above but with the dependent variable always log transformed.

For in situ parameters like pH or dissolved oxygen, EDA suggested different sonde models often had consistent and potentially different patterns across time, even after all error checks and calibration QAQC procedures were completed. As this might be another source of variation, like a lab effect in need of statistical control, we treat sonde type as a factor and include it and a sonde by scaled date interaction in all models of in situ responses. We refer to sonde and lab interchangeably as these are functionally the same in the QR models.

To visualize and estimate trend in a final model we extract partial residuals from the scaled date term (Larsen and McCleary 1972; Breheny and Burchett 2017). A partial residual is a dependent variable corrected for all independent variables except a variable of interest in the model. Plots of partial residuals show the relationship between a given independent variable and the response variable given that other independent variables are also in the model. The plot shows the value of the scaled date variable on the x-axis and the partial residual on the y-axis, holding all other variables constant (by default, median for numeric variables and most common category for factors). We include a quantile regression fit to the median in the partial plots. The coefficient (or slope) for this fit is the best or

corrected estimate of any trend—accounting for any effects of all other predictors in the model such as season or flow, and most importantly, for chemistry responses, lab and its interaction with scaled date. However, to avoid double hypothesis testing, the significance of any trend comes from the main model scaled date term.

QR results are presented in a familiar ANOVA format with coefficients, standard errors, and the significance of the term (p-value). We report the overall adjusted model R^2 as estimated for a QR by Koenker and Machado (1999) and the change in AIC over a null intercept only model. We used the rq function in the quantreg package (Koenker 2021) for all QR models. Model and data processing R code is available from the network, via IRMA and/or via github.

Censored trend models vs. imputation

A common feature of chemistry datasets is that one to many results reported from a lab can be below a detection capacity or limit of an analytical protocol. These are labeled as a non-detect or censored results. Chemists consider these results to have low reliability—hence the censoring. This topic has been evaluated in the literature for many years and is an important aspect of any protocol for the analysis of chemistry data (e.g., Helsel and Cohn 1988; Klein and Moeschberger 2003; Helsel 2005; Hirsch et al. 1991. To properly model censored data, the non-detected values must be dealt with.

Dated methods we discarded without considering included replacing censored values with the detection limit or some arbitrary fraction of a detection limit. However, this exhibits either extreme bias or much higher variability than other options (Helsel 2005). Two other solutions, Tobit regression (Helsel 1990) and censored quantile regression (Portnoy 2003), were evaluated extensively. Censored quantile regression was especially attractive given our use of QR for site-parameter datasets unaffected by censoring. However, the available tools for its implementation are complex and a black box without readily calculable features such as AIC, adjusted R^2, and most importantly for our purposes, partial residual extraction and interpretation. Tobit regression also has its merits and we used it in Schweiger et al. (2014a, b) and Mills et al. (2011) for Rocky Mountain Network trend models in three parks, including Glacier. However, the method may be biased, with high percentages of censored data. It is also a parametric model and thus subject to the same limitations presented above (Hirsch et al. 1991) that motivated our choice of QR for non-censored data.

Given these shortcomings, we replaced censored data with an estimated or imputed value and then conducted a quantile regression (as described above) of a combined dataset with raw or original data and imputed values in place of censored results. To impute data, we used Regression on Order Statistics (ROS; Shumway et al. 2002). ROS orders raw or measured values in a standard normal Quantile-Quantile plot, places censored values into this, and then predicts a replacement from this ordered dataset with a linear regression. ROS is a semi-parametric method in the sense that part of the inference is based on an assumed (normal) distribution and part of the inference is based on existing SEI data. ROS performs well with small sample sizes (the range of sample sizes used in our ROS models was 83 to 242, depending on the parameter, so it was not really all that small) and can accommodate multiple detection limits (common in our data). ROS works very well on small to moderate levels of censoring (up to 50%) and performs reasonably well with up to 80% percent of

values censored. We do not model trend for any parameter with over 80% censoring. Censoring thresholds are set based on the results for each parameter at a site and tend to increase in streams with very dilute or oligotrophic water. ROS was not applied to in situ parameters as there is no equivalent to censoring as done in lab-based results. The US EPA (EPA 2004) and the State of Colorado (CDPHE 2023) have incorporated ROS methods into environmental guidance documents. We also used the method for the 2007–2009 Glacier survey report (Schweiger et al. 2018) and thus several park median values reported here include ROS estimated values.

Sample size requirements for trend models

Trend models require a minimum period of record and overall sample size to be reliable (Table 32). If a season or lab term is included, there is also a minimum density of samples within each of these to estimate unbiased coefficients. Hem (1989) and Weissinger (2016) provide general guidance in setting the required sample size, sample duration, and sample density for trend models. Sample size and duration was calculated as the total number and range of years over which sampling occurred, respectively. Sample density was the number of years sampled over the range of years sampled. We also summarized the total number of labs used over the years of sampling for each site-parameter and the total number of unique detection limit (DL) and the minimum and maximum DL. Because we replace censored values using ROS, we don't need to calculate or limit models by the proportion of results censored, other than the upper limit of 80% noted above.

Table 32. Sample size and period of record requirements for quantile regression models.

Attribute	Minimum	Description and notes
Sample duration	5 or more years	Most site-parameter periods of record were around 10 years
Total sample size	10 or more	Based on 3*number of seasons, with a minimum of 10 (we only have samples in 3 seasons)
Density of samples	0.5	Proportion of years with samples
Minimum N per lab	8 [a]	With low variance, false positives and false negatives can occur at N < 8, but distributional form, (shape) is identifiable at N ≥ 8 (Jenkins and Quintana-Ascencio 2020)

[a] The minimum sample size per lab is smaller than ideal (~25), but this is what it is and we moved forward with analyses. We omit data from any lab with fewer than 8 samples.

For site-parameters that do not meet one or more of these criteria, we calculated the mean, standard deviation, median, minimum, and maximum values using ROS imputed values. This was always the case for sites North Fork Flathead—Glacier Rim and Middle Fork Flathead, where we began ongoing sampling in 2018. Samples from 2008 at site North Fork Flathead—Glacier Rim do not allow a trend model.

Habitat and biology

Most of the details presented above for chemistry are applicable to habitat and biological responses. We begin with EDA, assumption tests, and then apply quantile regression to model trend. However, there are no lab effects, nor any seasonal term given single samples per year. We explore the inclusion of covariates such as flow or water balance derived drainage but given sample size

constraints and complex structural relationships between variables, we only include these terms when there are compelling reasons to do so. Future work will explore models across sites or even across network parks.

Continuous responses: General Additive Models of water temperature

Modeling time-series can be tricky, as change over time in data that are continuous can be strongly non-linear and autocorrelated. Exploratory data analysis (EDA) of time series of water temperature suggested that General Additive Models (GAM) would be an ideal model form. Alternate methods such as linear regression have limitations with serially autocorrelated non-linear data and may therefore provide a poor description or prediction of pattern. We present our GAM analytical methods in detail here as this is the first time they have been published by the network.

Time and lags

We centered and normalized the daily maxima time series, rescaling days based on the mean, minimum, and maximum date sampled at each site. Scaling improves interpretation of model coefficients for time terms. We refer to time treated in this way as "Scaled Date."

We conducted exploratory work to determine the best time step over which to understand water temperature. We estimated draft GAM models (see below) using daily, weekly, and monthly intervals. To determine if there were possibly important lags in the data, we ran exploratory bivariate linear regressions to identify if a water balance predictor had a meaningful lagged relationship with water temperature over a range of 1 to 365 days. If any lagged adjusted R^2 was significantly better than a contemporaneous relationship, we created a lagged variable at this time step and included this as a new variable in the estimation of a GAM model. We limited lagged terms to a single instance (i.e., we only could have one lagged term for each water balance predictor), although there were often gradients in lagged effects.

General Additive Model details

To model water temperature, we use General Additive Models (GAM; Wood 2011) in which non-linear or "smooth" functions can be applied to all or a subset of predictor(s) as patterns in data indicate. Smooths are regularized nonparametric functions that can fit simple to very complex non-linear relationships. GAM models may also include predictors with a linear relationship with the response and/or with an a priori or expected non-linear relationships, such as a log. All terms in GAMs are estimated simultaneously and then the response predicted by adding these terms up. Smooths do not have to have classic polynomial form; for example, they can be complex "wiggly" fits. Moreover, we do not need to know the shape of a given smooth a priori as they are derived based on the data during model estimation. The smoothness of a predictor function can be controlled in a GAM to deal with a bias–variance tradeoff and avoid overly wiggly, nonsensical functions. Through the setting of smooth parameters, a GAM can include the prior belief that a given predictive relationships is inherently smooth even though often data suggest a noisy relationship. This plays an important role in model interpretation as well as in the believability of the results.

Using contemporaneous and lagged versions (if there was a meaningful lagged relationship) of the stream flow (if available), water balance metrics, air temperature, precipitation, and time-based

predictors day of year and season resulted in a large suite of candidate variables. We initially conducted model selection using a combination of hand and automated steps and four model selection criteria (in order of importance): 1) Akaike Information Criterion (AIC; Venables and Ripley 2002), 2) overall model and individual predictor interpretability, 3) adjusted R^2 values, and 4) a candidate predictor's p-value. However, these models were often quite complex and there was not a clear diagnostic or repeatable solution. We therefore reverted to a more structural approach following Grace et al. (2020) and limited predictors to those with likely more direct or readily explainable indirect connections to water temperature. We still employed a model selection process, but it was much simplified with the suite of predictors constrained.

We explored several alternate distributions for the GAM models but determined that a Gaussian distribution with a link identity function worked well and was simplest. Various shape and other parameters for smoothed terms were explored and set based on changes in our model selection criteria—in the end we used a thin plate spline method but let the basis number in these splines vary. Models were fitted via restricted maximum likelihood estimation.

GAM models assume that errors (residuals) are identically and independently distributed. However, like with any model that includes a temporal component, this is strong assumption—rarely fulfilled. If autocorrelation (i.e., water temperature in day one predicts water temperature in day two) is not addressed, estimated coefficients and residuals can be negatively biased and p-values or confidence intervals wrong. We therefore used the GAMM (General Additive *Mixed* Models) variant on GAMs that allows random predictors and/or an ARIMA (Auto Regressive Integrated Moving Average) component, both of which can deal with correlated residual structure. We determined if a GAMM was required by examining the residuals of candidate models via autocorrelation (ACF) and partial autocorrelation (pACF) function plots (Makridakis et al. 1998). If these suggested a strong pattern in the residuals, we used an automated function in the forecast package in R (Hyndman et al. 2023) to determine the best ARIMA dimensions and estimated a GAMM model with the indicated autoregressive and/or moving average lags. An ARIMA component in a model estimates corrected error terms, helping to ensure that model coefficients and standard errors are unbiased and correct. An ARIMA adjustment also can affect overfit (too wiggly) scaled date smooths in a candidate model, even suggesting replacement with a linear term. Linear scaled date terms address most concisely SEI monitoring objectives and this was a desirable outcome of the ARIMA adjustment. We also did this when the ARIMA adjusted scaled date smooth was similar to a linear alternative scaled date term. However, if the modeling process indicated that for a given site, trend in a response was non-linear and should not be changed, we fit the scaled date term with a smooth. Smooth terms do not have simple/single coefficients describing a simple trend over the period of record as the slope of fit is potentially different at every point in the time series. Where necessary we approximate an overall scaled date trend by estimating increasing or decreasing periods of change in the time series using a method of first derivative analysis (Monteith et al. 2014) as applied to GAMs in Simpson (2018). We set the period over which a derivative was estimated to a single day and estimated 95% point-wise confidence intervals on the standard errors of the derivatives. A confidence interval around a derivative that does not include zero indicates that, for that interval, the slope was significantly

different from zero. We summarize a mean derivative and the percent of the days across the period of record that do not include zero (and are thus significant).

To assist in the interpretation of variables in the final model, we estimated the relative contributions of each of the predictors to a model's total explanatory value. We used a classic (Grömping 2006) approach that averages sequential sums of squares over all possible orderings of regressors (Lindeman et al. 1980). This produces estimates of each predictor's importance relative to the total amount of explained variance expressed as a simple percentage. These estimates consider both direct effects (i.e., correlations with the response) and effects when combined with the other variables in the model. The method averages over all possible orderings of predictors in a model, thereby dealing with multicollinearity (Grömping 2006).

At all stages of modeling, we performed checks on the distribution of residuals, number of basis dimensions, multicollinearity of linear terms, and concurvity of smoothed predictors. Concurvity is the generalization of collinearity in a GAM where smooth terms can be approximated by a combination of other smooth terms and can result in unstable estimates. To reduce concurvity in our models, we removed variables from pairs with correlations > 0.70, omitting the variables with other issues and/or poorer interpretation. We also examined QQ plots of residuals and determined if the smoothing dimensions (k) was restrictively low using the gam.check function (Wood 2011).

GAM results are presented in a familiar ANOVA format with parametric term coefficients, standard errors, the significance of the term (p-value), and relative importance (grouped for factor predictors like season). For smoothed terms we report the estimated degrees of freedom (EDF) that indicate the degree of smoothing in a fit, the F value, and significance of the smoothed fit (p value). We use p-values as a model selection criterion (in part); however, we do not require a predictor to be significant to be included if AIC is improved and the term is generally interpretable (*sensu* Murtaugh 2014). Finally, we report the overall adjusted model R^2, the change in AIC over a null intercept only model, and the form of any ARIMA term used.

To visualize final model results we use plots of partial residuals (Larsen and McCleary 1972; Breheny and Burchett 2017). A partial residual plot is a graphical tool that shows the relationship between a given independent variable and the response variable, given that other independent variables are also in the model. The plot shows the value of the variable on the x-axis and the partial residual on the y-axis, holding all other variables constant (by default, median for numeric variables and most common category for factors).

We used the mgcv GAM and GAMM models and the corARIMA function from the nlme library for ARIMAs (Wood 2011). All model and data processing R code is available from the network, via IRMA and/or via github.

Appendix C: Continuous Water Temperature Trend Model Methods and Results

Trend model development

Water temperature can be summarized with a variety of metrics. We use a maximum daily temperature for trend analyses because climate change is increasing water temperature maxima the most dramatically and this metric might have the most important ecological implications (Rood et al. 2005; Muhlfeld et al. 2011, Jordan et al. 2016). Maximum temperatures may also have the most severe physiological implications for aquatic biota as most species' metabolic rates increase exponentially with temperature—but only to a point—and small temperature increase at the highest temperature can have relatively large physiological effects (Buckley and Huey 2016). Maxima are also what most regulatory criteria and other thresholds focus on.

We used General Additive Models (GAM) to estimate trend in maximum daily water temperature and to identify and describe predictors that might help interpret this important ecological response metric. Models used a model selection approach with maximum daily water temperature as the response and climate, water balance metrics, and stream flow as continuous predictor variables. Temporal predictors included a day of year (DOY), seasons, and time (scaled and normalized to range between 0 and 1). Because we did not model water temperatures across sites, spatially variable predictors, like elevation or local canopy cover, are not used. These types of predictors often explain large amounts of variation across space in water temperature (Jones et al. 2014, 2017; Isaak, Wenger, Peterson et al. 2017; Isaak et al. 2020).

Exploratory work often suggested a complex suite of climate and water balance predictors for most sites. Following Grace et al. (2020), we simplified models as much as possible, limiting predictors to those that more likely had a direct connection or indirect connection but with fewer latent variables in a causal connection (Briske et al. 2015; Sanders et al. 2017; Isaak, Luce, Chandler et al. 2018; Isaak, Luce, Horan et al. 2018). Terms in the GAMs were selected under hypotheses of a candidate variable being either drivers of water temperature, such as drainage or discharge (Isaak, Wenger, Peterson et al. 2017).

, 2020), or responses to water temperature such as (depending on the direction of the fit) evapotranspiration or soil moisture (Kaandorp et al. 2019), (or both). Our intent was to best predict temperature time series and estimate trend. Both structural drivers and responses to temperature dynamics contribute to this.

Testing for autocorrelation in a candidate model's residuals was a key step in the model estimation process. Initial GAM models included a significant and usually complex ("wiggly") smooth of water temperature across Scaled Date, with no or little visual suggestion of an overall trend across the period of record. However, model diagnostics suggested that these smooth terms were likely overfit and strongly driven by autocorrelation across daily maximum temperature readings. They therefore required an autoregressive integrated moving average (ARIMA) adjustment to account for this temporal structure. ARIMA adjustments of varying form controlled autocorrelation in the residuals

and (somewhat unexpectedly) allowed the heretofore complex nonlinear smoothed term for Scaled Date to be reduced to a linear fit. This allowed rigorous models with a traditional single coefficient and its p-value for trend. Parametric terms in a GAM have a traditional coefficient with an associated standard error and p-value, allowing a simpler statement regarding overall trend in daily maximum water temperature at each site (a core SEI objective).

Detailed trend model results

Final GAMs are in Table 33 with partial residual plots in Figures 81–85. We provide detailed interpretation to facilitate the understanding of trend in water temperature at these sites and how climate, water balance, and hydrology might matter. Overall, there was a high degree of similarity across the models, suggesting meaningful generality in our estimates of trend in water temperature. All models explained a large portion of the variation in maximum daily water temperature at each site and included interpretable water balance and/or hydrologic drivers.

Table 33. General Additive Models (GAM) of weekly maximum water temperature at SEI sentinel sites in Glacier National Park, 2007–2019. We report parametric term (P) coefficients (Coeff.), standard errors (SE), the significance of the term (p) and relative importance (RI; grouped for factor predictors like season). For smoothed terms (S) we report the estimated degrees of freedom (EDF) that indicate the degree of smoothing in a fit, the F value, and significance of the smoothed fit (p). The final model adjusted R^2 indicates the overall proportion of total variation in the response explained by the model. The improvement in fit over a null model is given by Delta AIC. The autoregressive model (AR) and/or moving average (MA) model parameters indicate if an Autoregressive or Moving Average (respectively) was used to account for autocorrelation with the optimized numerical order for each term as indicated.

Site	Term type	Term	Coeff.	SE	t value	EDF	F value	p	RI	Adjusted R^2	ARIMA
Swiftcurrent Creek	P	Intercept	6.740	1.935	3.484	–	–	**0.001** [a]	–	0.828	AR(3) MA (1)
	P	Scaled Date	0.279	0.111	2.510	–	–	**0.012** [a]	0.027	Delta AIC over null	–
	P	Season: Summer (vs. Spring)	0.338	0.355	0.952	–	–	0.341	0.219	5059.904	–
	P	Season: Fall (vs. Spring)	0.422	0.514	0.821	–	–	0.412	0.219	–	–
	P	log(Drainage)	−1.128	0.396	−2.845	–	–	**0.004** [a]	0.143	–	–
	P	GDD, 8-day lag	0.111	0.012	9.114	–	–	**0.000** [a]	0.379	–	–
	P	Soil Moisture	−0.004	0.009	−0.492	–	–	0.623	0.213	–	–
	S	s(Day of Year)	–	–	–	7.163	35.710	**0.000** [a]	0.019	–	–
Reynolds Creek	P	Intercept	10.470	3.092	3.386	–	–	**0.001** [a]	–	0.761	AR(2) MA (2)
	P	Scaled Date	0.168	0.153	1.097	–	–	0.273	0.024	Delta AIC over Null	–
	P	log(Drainage)	−2.432	0.904	−2.690	–	–	**0.007** [a]	0.526	2604.707	–
	S	s(Day of Year)	–	–	–	5.832	12.320	**0.000** [a]	0.450	–	–
Ole Creek	P	Intercept	9.239	0.692	13.355	–	–	**0.000** [a]	–	0.826	AR(2)
	P	Scaled Date	0.057	0.033	1.719	–	–	**0.086** [a]	0.016	Delta AIC over null	–
	P	Season: Summer (vs. Spring)	0.642	0.394	1.631	–	–	0.103	0.170	2432.713	–

[a] Term is significant ($p \leq 0.10$)

Table 33 (continued). General Additive Models (GAM) of weekly maximum water temperature at SEI sentinel sites in Glacier National Park, 2007–2019. We report parametric term (P) coefficients (Coeff.), standard errors (SE), the significance of the term (p) and relative importance (RI; grouped for factor predictors like season). For smoothed terms (S) we report the estimated degrees of freedom (EDF) that indicate the degree of smoothing in a fit, the F value, and significance of the smoothed fit (p). The final model adjusted R^2 indicates the overall proportion of total variation in the response explained by the model. The improvement in fit over a null model is given by Delta AIC. The autoregressive model (AR) and/or moving average (MA) model parameters indicate if an Autoregressive or Moving Average (respectively) was used to account for autocorrelation with the optimized numerical order for each term as indicated.

Site	Term type	Term	Coeff.	SE	t value	EDF	F value	p	RI	Adjusted R^2	ARIMA
Ole Creek (continued)	P	Season: Fall (vs. Spring)	0.742	0.574	1.292	–	–	0.197	0.170	–	–
	P	GDD, 7-day lag	0.055	0.013	4.312	–	–	**0.000** [a]	0.211	–	–
	P	log(Drainage)	−1.831	0.202	−9.042	–	–	**0.000** [a]	0.307	–	–
	P	Soil Moisture	−0.003	0.006	−0.403	–	–	0.687	0.276	–	–
	S	s(Day of Year)	–	–	–	6.491	36.270	**0.000** [a]	0.020	–	–
North Fork Flathead—Border	P	Intercept	20.346	0.846	24.037	–	–	**0.000** [a]	–	0.843	AR(4) MA (5)
	P	Scaled Date	0.012	0.006	1.792	–	–	**0.073** [a]	0.004	Delta AIC over null	–
	P	log(Discharge)	−1.523	0.126	−12.101	–	–	**0.000** [a]	0.175	8545.593	–
	P	Actual ET, 10-day lag	0.223	0.049	4.567	–	–	**0.000** [a]	0.393	–	–
	P	GDD	0.034	0.008	4.289	–	–	**0.000** [a]	0.362	–	–
	S	s(Day of Year)	–	–	–	7.353	165.000	**0.000** [a]	0.066	–	–
North Fork Flathead—Glacier Rim	P	Intercept	26.529	1.115	23.794	–	–	**0.000** [a]	–	0.872	AR(2) MA (3)
	P	Scaled Date	−0.006	0.014	−0.401	–	–	0.688	0.001	Delta AIC over null	–
	P	Season: Summer (vs. Spring)	0.333	0.169	1.970	–	–	**0.049** [a]	0.229	11423.200	–
	P	Season: Fall (vs. Spring)	0.099	0.239	0.416	–	–	0.678	0.229	–	–

[a] Term is significant ($p \leq 0.10$)

Table 33 (continued). General Additive Models (GAM) of weekly maximum water temperature at SEI sentinel sites in Glacier National Park, 2007–2019. We report parametric term (P) coefficients (Coeff.), standard errors (SE), the significance of the term (p) and relative importance (RI; grouped for factor predictors like season). For smoothed terms (S) we report the estimated degrees of freedom (EDF) that indicate the degree of smoothing in a fit, the F value, and significance of the smoothed fit (p). The final model adjusted R^2 indicates the overall proportion of total variation in the response explained by the model. The improvement in fit over a null model is given by Delta AIC. The autoregressive model (AR) and/or moving average (MA) model parameters indicate if an Autoregressive or Moving Average (respectively) was used to account for autocorrelation with the optimized numerical order for each term as indicated.

Site	Term type	Term	Coeff.	SE	t value	EDF	F value	p	RI	Adjusted R^2	ARIMA
North Fork Flathead—Glacier Rim (continued)	P	log(Discharge)	−1.856	0.137	−13.502	–	–	**0.000** [a]	0.112	–	–
	P	Actual ET, 20-day lag	0.002	0.031	0.051	–	–	0.959	0.275	–	–
	P	GDD, 7-day lag	0.049	0.005	9.359	–	–	**0.000** [a]	0.356	–	–
	S	s(Day of Year)	–	–	–	7.736	150.700	**0.000** [a]	0.027	–	–

[a] Term is significant ($p \leq 0.10$)

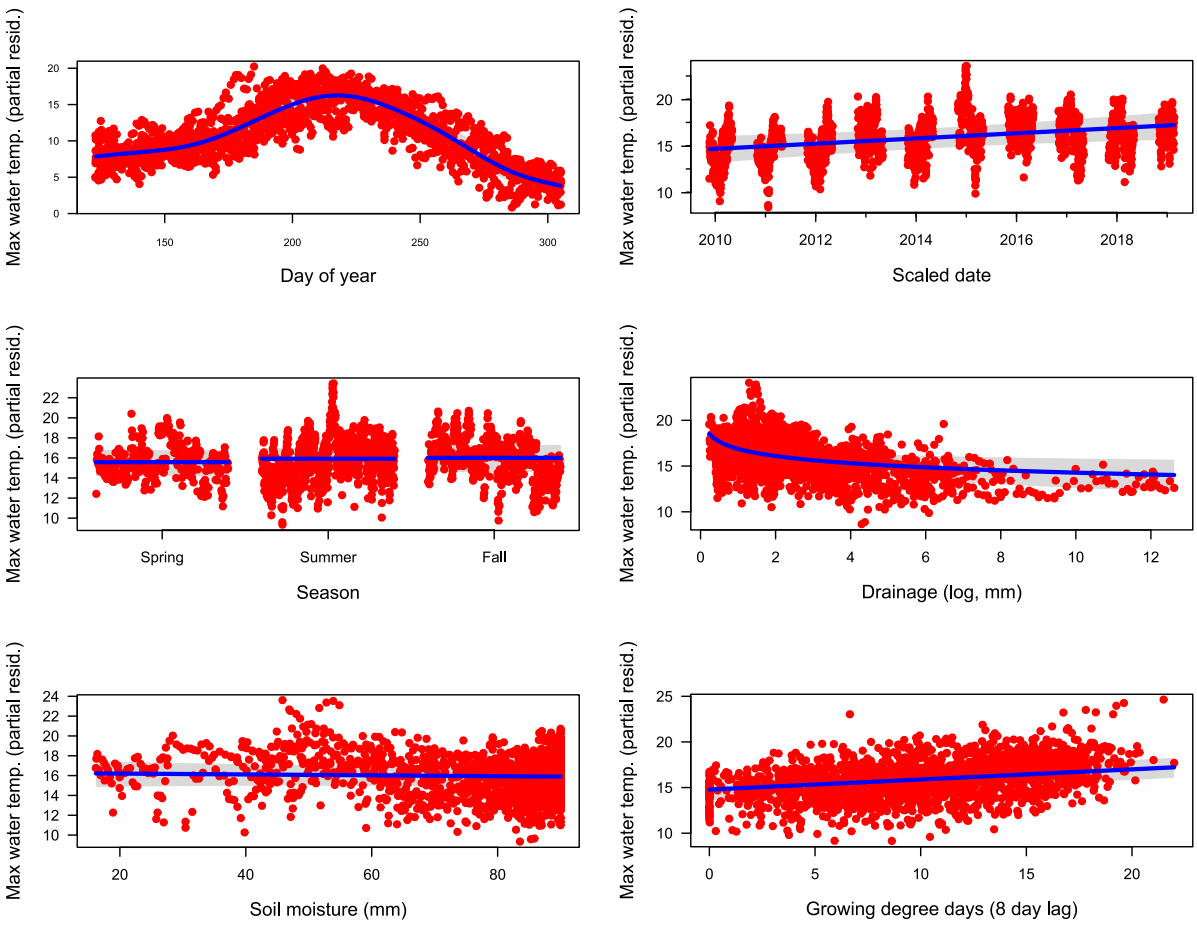


Figure 81. Partial residual plots for the final Auto Regressive Integrated Moving Average (ARIMA) adjusted General Additive Mixed Model (GAMM) models of daily maximum water temperature at Swiftcurrent Creek in Glacier National Park. Each panel shows the relationship between terms in a model and the partial residual of maximum daily water temperature, given that other independent variables are also in the model. Gray shaded areas give the confidence interval around each fit of the partial residuals for each term.

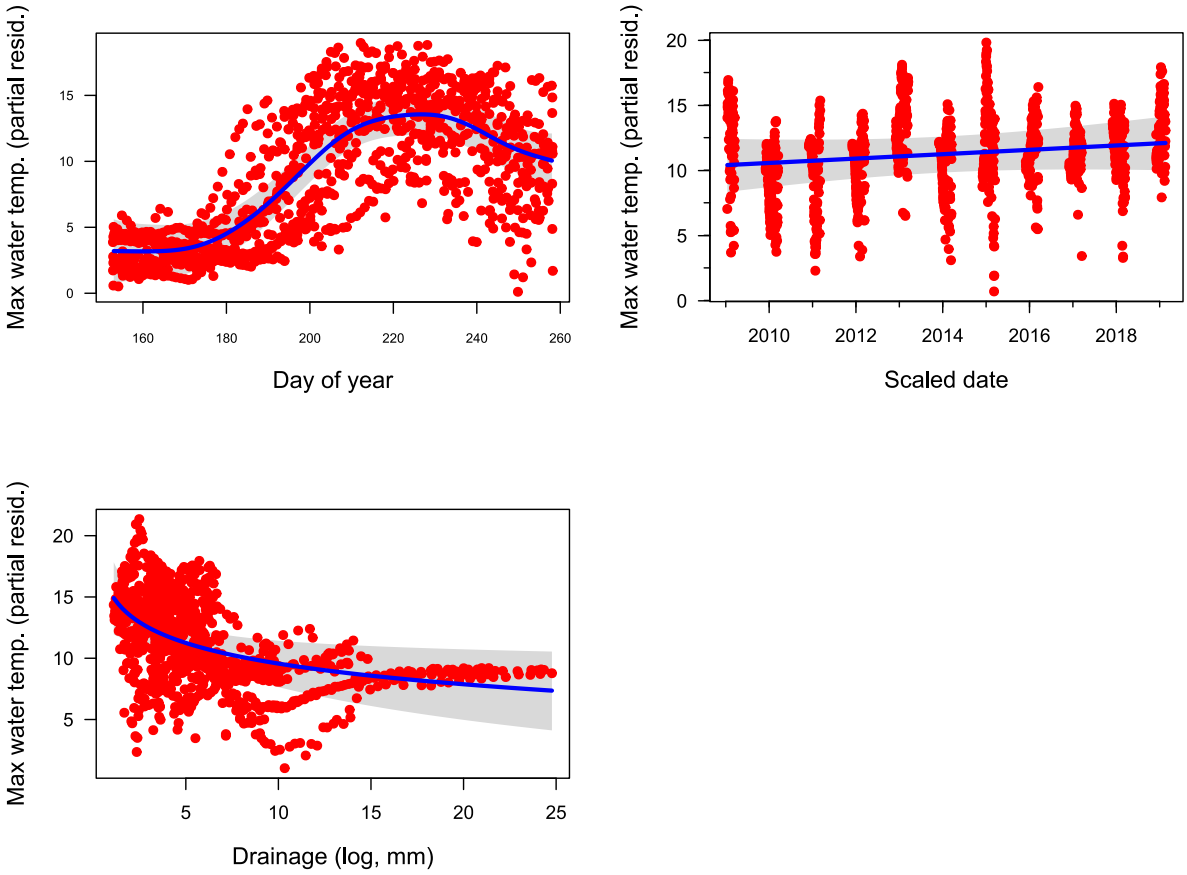


Figure 82. Partial residual plots for the final Auto Regressive Integrated Moving Average (ARIMA) adjusted General Additive Mixed Model (GAMM) models of daily maximum water temperature at Reynolds Creek in Glacier National Park. Each panel shows the relationship between terms in a model and the partial residual of maximum daily water temperature, given that other independent variables are also in the model. Gray shaded areas give the confidence interval around each fit of the partial residuals for each term.

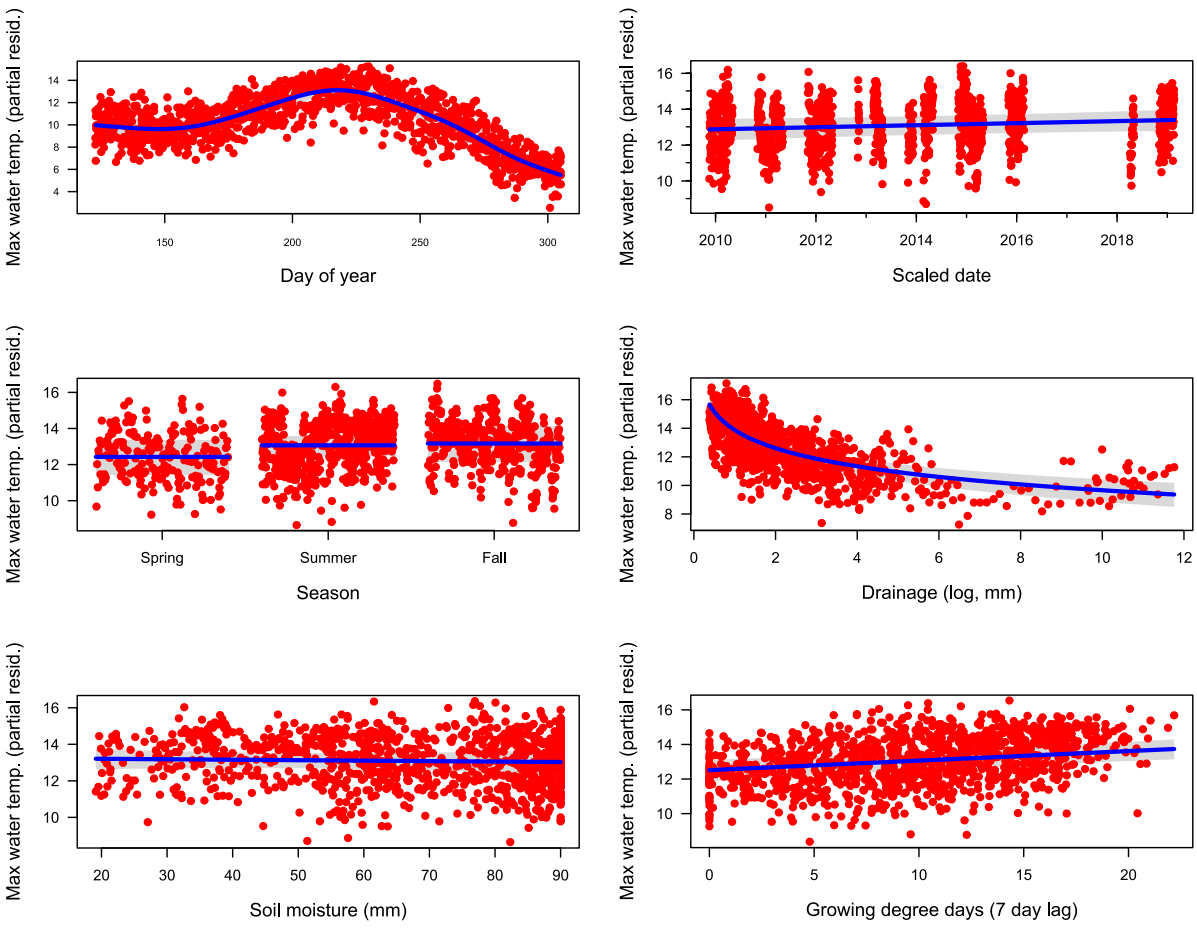


Figure 83. Partial residual plots for the final Auto Regressive Integrated Moving Average (ARIMA) adjusted General Additive Mixed Model (GAMM) models of daily maximum water temperature at Ole Creek in Glacier National Park. Each panel shows the relationship between terms in a model and the partial residual of maximum daily water temperature, given that other independent variables are also in the model. Gray shaded areas give the confidence interval around each fit of the partial residuals for each term.

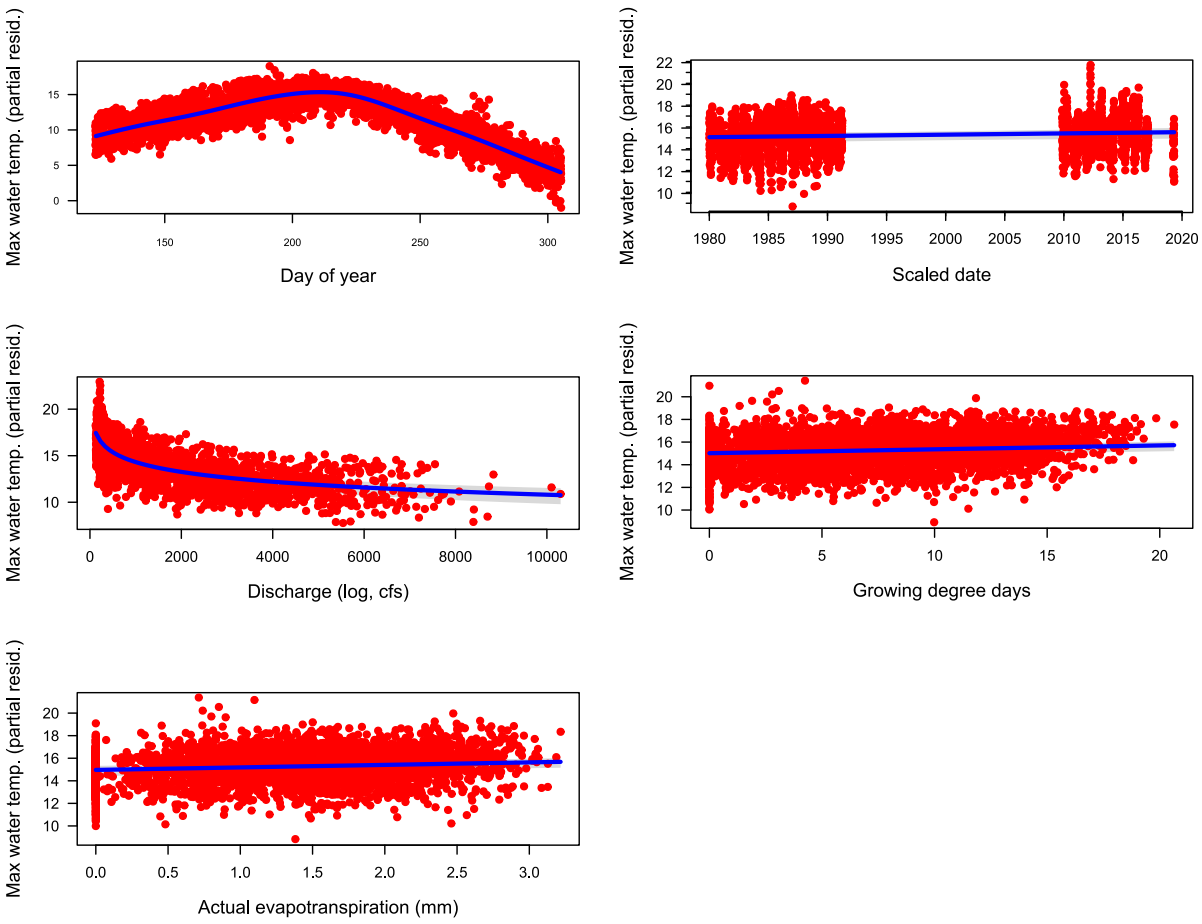


Figure 84. Partial residual plots for the final Auto Regressive Integrated Moving Average (ARIMA) adjusted General Additive Mixed Model (GAMM) models of daily maximum water temperature at the North Fork Flathead—Border stream monitoring site in Glacier National Park. Each panel shows the relationship between terms in a model and the partial residual of maximum daily water temperature, given that other independent variables are also in the model. Gray shaded areas give the confidence interval around each fit of the partial residuals for each term.

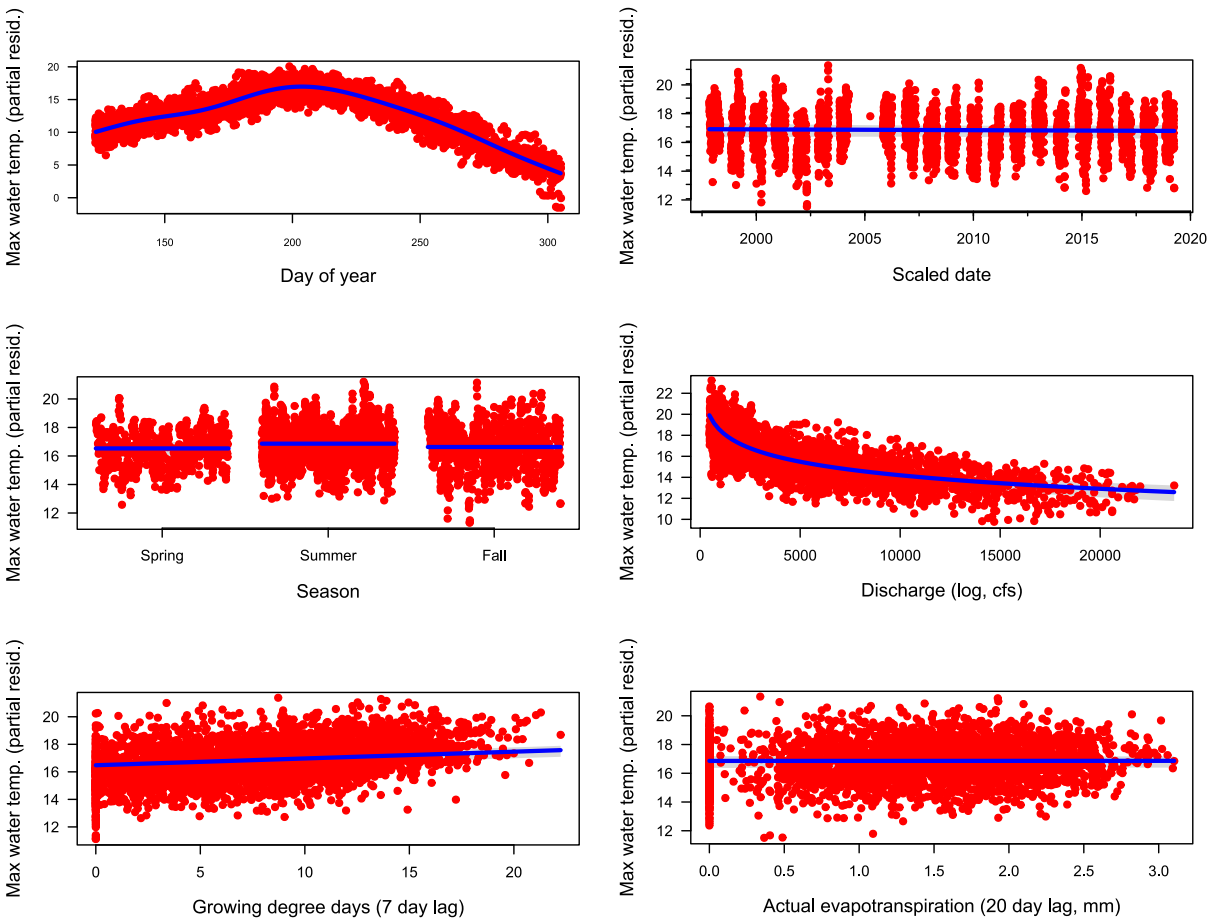


Figure 85. Partial residual plots for the final Auto Regressive Integrated Moving Average (ARIMA) adjusted General Additive Mixed Model (GAMM) models of daily maximum water temperature at the North Fork Flathead—Glacier Rim stream monitoring site in Glacier National Park. Each panel shows the relationship between terms in a model and the partial residual of maximum daily water temperature, given that other independent variables are also in the model. Gray shaded areas give the confidence interval around each fit of the partial residuals for each term.

Maximum daily water temperature increased at four out of five sites and these trends were significant or nearly so. There was a remarkable increase of about a 1 to 1.5 °C rise in temperature per decade at the four sites, with meaningful trends. The relative importance of Scaled Date was generally low, suggesting that while time was important in water temperature dynamics at many of our sites, trend across time explained less of the overall variation than other time related terms (season and day of year) or the water balance/hydrologic predictors in the models. All models included a strongly significant non-linear smoothed term for day of year (DOY based on water year) with a characteristic bell-shaped fit. This was likely the most important impact of using GAM models, as otherwise this term would have been difficult to fit. Season (as a factor or categorical term) was also in most models. Patterns across season were intuitive, with highest temperatures in the summer, although these differences were rarely significant. Partial residual plots (Figures 81–85) show the pattern or effect of each term in a model when all other terms are controlled for. These plots are the best way to

visually interpret complex multivariate models such as GAMs. However, given large sample sizes and a variable response (typical of ecological processes), there is a great deal of variation around each fitted term.

From one to three climate or water balance terms (see Appendix B) appeared in each model. These were consistently some combination of growing degree days (GDD), drainage or discharge, soil moisture, and actual evapotranspiration. Drainage was in three of the five models, somewhat surprisingly in place of continuous stream flow at Swiftcurrent Creek, where we had both these candidate variables. All three of these watersheds (Swiftcurrent, Reynolds, and Ole Creeks) are relatively small compared to the two big river sites where continuous flow outperformed drainage in modeling maximum water temperature. Drainage and stream flow are strongly related, and both are hydrologic drivers; however, given the drainage term's calculation from water balance-based runoff, it may include more of a groundwater component, especially in the smaller watershed and gaining stream reaches typical of our three smaller stream sites. Groundwater can have a strong impact on water temperatures (Kaandorp et al. 2019) and the improvements in the three smaller watershed models from including drainage may reflect this. Drainage may also be better connected to water temperature in non-baseflow seasons because of its water balance basis. Drainage was also the most important predictor in two of these three models, sometimes accounting for more than half of the explained variation in maximum temperatures. This suggests a strong connection between the water cycle and water temperature, with important climate change implications. The ARIMA adjustments in each model allowed drainage or discharge to be fit as a parametric logged term, allowing a trend estimate from a traditional coefficient. Logged fits of drainage or discharge to maximum temperature outperformed linear and even smoothed fits, suggesting a consistent exponential relationship between hydrology and water temperature. At all sites, the logs of drainage and maximum water temperatures were negatively related with an increasing rate of increase in maximum water at lower drainage values, often in later summer at base flow. Warming water may lead to more and more warming in a water body. The two sites where stream flow was selected over drainage, North Fork Flathead—Border and North Fork Flathead—Glacier Rim, are both on the North Fork of the Flathead, with large watersheds with complex landcover, variable terrain, and other complicating factors. There may be less relative groundwater contribution to the sites given their size, perhaps explaining the inclusion of discharge over drainage, but this needs more work.

Growing degree days (GDD) was also a key linear predictor of maximum water temperatures at four of five sites. It had an intuitive positive relationship and the highest or second highest importance in all models except at the alpine site, Reynolds Creek. Three of the four GDD terms were lagged to around a week, suggesting changes in warmth in the atmosphere take some time to translate to changes in warming waters—the well described hysteresis between air and water temperatures (Letcher et al. 2016). GDD is derived from air temperature, a classic term in models of water temperature, but GDD cannot be negative, while air temperatures can drop well below zero (Benyahya et al. 2007). Flowing water temperature also does not drop below zero in most cases, and this may be why GDD better predicts water temperature than air temperature does. It is not clear why GDD (or air temperature) did not appear in the Reynolds Creek model. The open canopy on the Reynolds Creek site allows daytime summer water temperature maxima to be quite high but

nighttime temperatures are very cold still. The ice-free season at this site is also shorter than at others, reducing the length of time over which a covariance structure between GDD and water temperature might develop. Both these may explain why GDD did not appear in the model, but this will need more work.

Soil moisture appeared in the models for Swiftcurrent Creek and Ole Creek, showing a relatively weak and nonsignificant negative relationship with maximum water temperature. These are the larger of the three small watershed sites in our dataset. They both contain relatively variable landcover and topology. Soil moisture is related to hydrologic processes in watersheds. For example, Harpold et al. (2017) demonstrate an improved prediction of stream flow when soil moisture is included in models. Lower soil moisture increases the potential to store precipitation or snowmelt within the soil profile, supporting runoff, whereas higher soil moisture can more easily exceed water holding capacity of soils in a watershed, causing drainage out of the soil profile and also influencing runoff and stream flow. Because stream flow and often runoff have a direct mechanistic connection to water temperature, this might explain the soil moisture term in our models. The effect is weak given the many latent connections between drainage, runoff, and water temperature. Yet it was relevant enough to get picked up in our model selection process.

Finally, actual evapotranspiration (AET) was in the models for the North Fork Flathead—Border and North Fork Flathead—Glacier Rim sites, with an important and positive relationship to maximum water temperature. In smaller more closed aquatic systems (i.e., streams), there is often a negative relationship between AET and water temperature, given simple evaporative cooling of a water body with increased AET (Kadlec 2006). We see positive and lagged relationships with AET in the two big river sites, suggesting that warmer water temperatures covary with increased AET. AET may also reflect increased, or even create, loss of water, lowering water depths, which generally elevates temperatures (Lupon et al. 2016). AET may also operate differentially on different limbs of the hydrograph, increasing temperatures on the rising limb but decreasing them on a falling limb (Cozzetto et al. 2006; Webb et al. 2008). In general, the form of this relationship (as a structural driver or as a response) is not entirely clear. Regardless, at these two sites, inclusion of AET improved the final model and our estimate and interpretation of trend in water temperature.

Appendix D: Taxa Lists

The following tables list taxa collected from 2007 to 2019 at all stream monitoring sample events at Glacier National Park sentinel sites.

Macroinvertebrates

Table 34. Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Ameletidae	*Ameletus*	100996	Genus	1256	3.16
	Ameletus bellulus	568565	Species	14	2.65
	Ameletus celer	101019	Species	102	16.67
Apataniidae	*Apatania*	115935	Genus	294	1.51
	Apataniidae	598182	Family	3	0.44
	Pedomoecus sierra	115973	Species	1	0.16
Athericidae	*Atherix*	130929	Genus	173.5	1.15
Aturidae	*Aturus*	82974	Genus	4	0.33
Baetidae	*Acentrella*	100801	Genus	298	2.32
	Anafroptilum	–	Genus	5	0.26
	Baetidae	100755	Family	176	2.76
	Baetis	100800	Genus	1123	6.26
	Baetis bicaudatus	100823	Species	122	2.16
	Baetis bicaudatus complex	–	Species complex	36	0.69
	Baetis flavistriga	100835	Species	116	2.01
	Baetis flavistriga complex	–	Species complex	41	0.88
	Baetis Fuscatus Gr.	–	Species group	31	4.64
	Baetis piscatoris complex	–	Species complex	27	1.38
	Baetis Rhodani Gr.	–	Species group	176	2.63
	Baetis tricaudatus	100817	Species	456	4.02
	Baetis tricaudatus complex	–	Species complex	1595	6.54
	Diphetor hageni	568598	Species	164	0.93
Blephariceridae	*Agathon*	121230	Genus	2	0.28
	Blephariceridae	121227	Family	2	0.17

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Brachycentridae	*Amiocentrus aspilus*	116934	Species	1	0.15
	Brachycentridae	116905	Family	7	0.60
	Brachycentrus	116906	Genus	64	0.66
	Brachycentrus americanus	116912	Species	83	0.59
	Brachycentrus occidentalis	116918	Species	4	0.30
	Micrasema	116958	Genus	75	0.77
Capniidae	*Capniidae*	102643	Family	173	0.71
Ceratopogonidae	*Ceratopogonidae*	127076	Family	9	0.46
	Ceratopogoninae	127338	Subfamily	448	1.70
	Forcipomyia	127152	Genus	1	0.19
	Mallochohelea	127702	Genus	33	1.75
Chironomidae	*Ablabesmyia*	128079	Genus	1	0.14
	Bilyjomyia algens	–	Species	1	0.16
	Boreochlus	127954	Genus	1	0.15
	Boreoheptagyia	128343	Genus	1	0.15
	Brillia	128477	Genus	40	0.26
	Cardiocladius	128511	Genus	3	0.18
	Chaetocladius	128520	Genus	43	0.46
	Chironomidae	127917	Family	1187	11.59
	Chironominae	129228	Subfamily	4	0.50
	Chironomini	129229	Tribe	76	0.69
	Cladotanytarsus	129873	Genus	77	1.03
	Constempellina	129884	Genus	2	0.15
	Corynoneura	128563	Genus	43	0.25
	Cricotopus	128575	Genus	378	1.17
	Cricotopus / Orthocladius	–	Genus	532	4.42
	Cryptochironomus	129368	Genus	1	0.14
	Diamesa	128355	Genus	156	0.89
	Diamesinae	128341	Subfamily	35	3.54
	Dicrotendipes	129428	Genus	2	0.15
	Diplocladius cultriger	128671	Species	1	0.16
	Eukiefferiella	128689	Genus	686	1.26
	Heleniella	128730	Genus	24	0.44

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Chironomidae (continued)	*Helopelopia*	128131	Genus	22	0.38
	Heterotanytarsus	128734	Genus	1	0.15
	Heterotrissocladius	128737	Genus	19	0.34
	Hydrobaenus	128750	Genus	86	0.98
	Krenopelopia	128170	Genus	2	0.15
	Krenosmittia	128771	Genus	15	0.29
	Limnophyes	128776	Genus	1	0.15
	Micropsectra	129890	Genus	1053	1.83
	Microtendipes	129535	Genus	231	1.14
	Monodiamesa	128440	Genus	3	0.23
	Nanocladius	128844	Genus	5	0.18
	Nilotanypus	128202	Genus	2	0.16
	Odontomesa	128446	Genus	2	0.16
	Orthocladiinae	128457	Subfamily	238	0.79
	Orthocladiinae sp. (RAI Taxon # 0004)	–	Species	8	0.42
	Orthocladiinae sp. (RAI Taxon # 0011)	–	Species	39	7.39
	Orthocladius	128874	Genus	856	1.66
	Orthocladius lignicola	128913	Species	3	0.15
	Pagastia	128401	Genus	529	1.44
	Parachaetocladius	128951	Genus	1	0.16
	Paracladopelma	129597	Genus	3	0.30
	Parakiefferiella	128968	Genus	269	1.81
	Parametriocnemus	128978	Genus	186	0.87
	Paratanytarsus	129935	Genus	14	0.20
	Parorthocladius	129011	Genus	4	0.31
	Phaenopsectra	129637	Genus	28	0.50
	Polypedilum	129657	Genus	329	1.32
	Potthastia	128408	Genus	52	0.67
	Procladius	128277	Genus	4	0.40
	Prodiamesa	128452	Genus	1	0.15
	Psectrocladius	129018	Genus	1	0.16
	Pseudochironomus	129851	Genus	12	0.92

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Chironomidae (continued)	*Pseudodiamesa*	128416	Genus	6	0.25
	Pseudorthocladius	129052	Genus	7	0.54
	Radotanypus	–	Genus	1	0.15
	Rheocricotopus	129086	Genus	233	0.64
	Rheopelopia	128226	Genus	5	0.24
	Rheosmittia	129107	Genus	1	0.17
	Rheotanytarsus	129952	Genus	150	0.67
	Robackia	129730	Genus	9	0.35
	Stempellina	129962	Genus	3	0.29
	Stempellinella	129969	Genus	825	1.41
	Stictochironomus	129785	Genus	14	0.49
	Stilocladius	129152	Genus	1	0.15
	Sublettea	129975	Genus	5	0.45
	Sublettea coffmani	129976	Species	188	1.80
	Synorthocladius	129161	Genus	28	0.28
	Tanypodinae	127994	Subfamily	56	0.64
	Tanytarsini	129872	Tribe	309	1.35
	Tanytarsus	129978	Genus	1014	2.97
	Thienemanniella	129182	Genus	36	0.36
	Thienemannimyia Gr.	–	Genus group	386	1.26
	Tvetenia	129197	Genus	705	1.36
	Zavrelimyia	128259	Genus	4	0.20
Chloroperlidae	*Alloperla*	103203	Genus	59	1.31
	Chloroperlidae	103202	Family	500	2.57
	Kathroperla	103236	Genus	1	0.16
	Paraperla	103233	Genus	184	1.24
	Suwallia	103254	Genus	380	2.01
	Suwalliini	–	Tribe	64	0.89
	Sweltsa	103273	Genus	3799	7.31
Corixidae	*Corixidae*	103364	Family	1	0.14
Crangonyctidae	*Crangonyx*	95081	Genus	1	0.16
Deuterophlebiidae	*Deuterophlebia*	121287	Genus	5	0.19
Dolichopodidae	*Dolichopodidae*	136824	Family	3	0.16

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Dytiscidae	*Dytiscidae*	111963	Family	8	0.35
	Hydroporinae	678402	Subfamily	2	0.16
	Laccornis	112270	Genus	1	0.19
	Liodessus affinis complex	–	Species complex	4	0.25
	Oreodytes	112314	Genus	24	0.34
	Oreodytes obesus	112316	Species	10	0.50
	Stictotarsus	568826	Genus	1	0.16
Elmidae	*Cleptelmis addenda*	114166	Species	8	0.24
	Elmidae	114093	Family	20	0.39
	Heterlimnius	114167	Genus	134	1.37
	Heterlimnius corpulentus	114169	Species	185	0.83
	Narpus	114142	Genus	4	0.35
	Narpus concolor	114144	Species	13	0.24
	Optioservus	114177	Genus	19	0.30
	Optioservus quadrimaculatus	114180	Species	2	0.30
	Ordobrevia	114235	Genus	2	0.36
	Zaitzevia	114205	Genus	443	1.95
	Zaitzevia parvula	945801	Species	6	0.23
	Zaitzevia posthonia	945813	Species	9	0.46
Empididae	*Chelifera*	136305	Genus	124	0.83
	Clinocera	135849	Genus	29	0.30
	Clinocerinae	135844	Subfamily	2	0.15
	Empididae	135830	Family	61	0.55
	Empididae sp. (RAI Taxon # 0001)	–	Species	3	0.24
	Hemerodromia	136327	Genus	10.5	0.65
	Neoplasta	136352	Genus	134	0.99
	Oreogeton	136377	Genus	178	1.59
	Roederiodes	135893	Genus	2	0.15
	Trichoclinocera	135903	Genus	15	0.38
	Wiedemannia	135920	Genus	15	0.48
Enchytraeidae	*Cernosvitoviella*	68534	Genus	1	0.16
	Chamaedrilus	–	Genus	9	0.46

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Enchytraeidae (continued)	*Enchytraeidae*	68510	Family	20	0.62
	Enchytraeus	68531	Genus	85	0.86
	Fridericia	204785	Genus	9	0.21
	Henlea	68541	Genus	2	0.30
	Mesenchytraeus	68544	Genus	32	0.61
Ephemerellidae	*Attenella margarita*	101343	Species	45	1.52
	Caudatella	101347	Genus	13	0.27
	Caudatella hystrix	101348	Species	1	0.12
	Drunella	101365	Genus	192	1.83
	Drunella coloradensis	101389	Species	312	1.37
	Drunella doddsi	101368	Species	140	2.00
	Drunella doddsii	698494	Species	1470	4.45
	Drunella flavilinea	101392	Species	35	1.87
	Drunella grandis	101370	Species	98	1.15
	Drunella spinifera	101385	Species	1	0.17
	Ephemerella	101233	Genus	881	3.27
	Ephemerella excrucians	101276	Species	158	3.49
	Ephemerella Excrucians Gr.	–	Genus	93	4.78
	Ephemerella inermis	101239	Species	89	7.93
	Ephemerella tibialis	101401	Species	142	0.74
	Ephemerellidae	101232	Family	22	0.44
	Serratella	101395	Genus	74	2.14
	Serratella tibialis	101399	Species	40	0.94
	Timpanoga hecuba	101318	Species	6	0.23
Ephemeridae	*Ephemera*	101526	Genus	53	4.75
Feltriidae	*Feltria*	83314	Genus	2	0.15
Glossosomatidae	*Glossosoma*	117159	Genus	547	2.03
	Glossosomatidae	117120	Family	35	0.51
Haliplidae	*Brychius*	111947	Family	5	0.77
	Haliplidae	111857	Family	1	0.15
Heptageniidae	*Cinygma*	100598	Genus	1	0.16
	Cinygmula	100557	Genus	1862	4.47
	Ecdyonurus criddlei	698149	Species	1	0.15

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Heptageniidae (continued)	*Epeorus*	100626	Genus	991	3.47
	Epeorus albertae	100629	Species	8	0.71
	Epeorus deceptivus	100632	Species	1214	4.75
	Epeorus grandis	100635	Species	187	1.42
	Epeorus longimanus	100637	Species	78	1.09
	Heptageniidae	100504	Family	1345	5.64
	Nixe	100692	Genus	1	0.18
	Rhithrogena	100572	Genus	5930	11.46
Hyalellidae	*Hyalella*	94025	Genus	7	0.25
Hydrophilidae	*Hydrophilidae*	112811	Family	2	0.11
Hydropsychidae	*Arctopsyche*	115529	Genus	313	1.40
	Arctopsyche grandis	115530	Species	139	1.75
	Arctopsychinae	568763	Subfamily	13	0.50
	Hydropsyche	115453	Genus	613	2.94
	Hydropsyche Morosa Gr.	–	Species group	33	1.69
	Hydropsychidae	115398	Family	167	1.22
	Parapsyche	115556	Genus	14	0.48
	Parapsyche elsis	115560	Species	22	0.27
Hydroptilidae	*Agraylea*	115635	Genus	1	0.17
	Hydroptila	115641	Genus	407	5.18
	Hydroptilidae	115629	Family	5	0.42
	Ochrotrichia	115714	Genus	1	0.15
Hydryphantidae	*Albertathyas montana*	–	Species	4	0.19
	Hydryphantidae	83212	Family	2	0.31
	Protzia	83170	Genus	58	0.41
Hygrobatidae	*Atractides*	83282	Genus	106	0.54
	Hygrobates	83297	Genus	295	1.70
	Hygrobatidae	83281	Family	4	0.72
Lebertiidae	*Lebertia*	83034	Genus	325	1.03
Lepidostomatidae	*Lepidostoma*	116794	Genus	287	1.03
Leptoceridae	*Ceraclea*	116684	Genus	9	0.28
	Leptoceridae	116547	Family	1	0.18
	Mystacides	116598	Genus	1	0.15

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Leptoceridae (continued)	*Nectopsyche*	116651	Genus	1	0.15
	Oecetis Avara Gr.	–	Species group	1	0.15
Leptohyphidae	*Tricorythodes*	101405	Genus	4	0.32
Leptophlebiidae	*Leptophlebiidae*	101095	Family	156	0.94
	Neoleptophlebia	–	Genus	1	0.14
	Paraleptophlebia	101187	Genus	290	1.44
	Paraleptophlebia bicornuta	101206	Species	47	0.81
	Paraleptophlebia Bicornuta Gr.	–	Genus	10	0.30
Leuctridae	*Despaxia augusta*	102842	Species	73	3.71
	Leuctridae	102840	Family	561	2.41
	Paraleuctra	102887	Genus	32	0.62
Limnephilidae	*Chyrandra centralis*	116018	Species	1	0.16
	Dicosmoecus atripes	116266	Species	3	0.16
	Dicosmoecus gilvipes	116268	Species	5	0.19
	Limnephilidae	115933	Family	229	2.04
	Psychoglypha	115974	Genus	139	2.21
Lumbriculidae	*Lumbriculidae*	68440	Family	27	0.42
	Rhynchelmis	68463	Genus	14	0.45
Lymnaeidae	*Lymnaeidae*	76483	Family	4	0.16
	Stagnicola	76534	Genus	6	0.32
Naididae	*Chaetogaster*	68934	Genus	1	0.16
	Limnodrilus	68638	Genus	1	0.15
	Naididae	68854	Family	11	0.19
	Naididae (Tubificinae) — with capillary setae	–	Subfamily	2	0.24
	Naididae (Tubificinae) — without capillary setae	–	Subfamily	11	0.60
	Nais	68946	Genus	217	1.62
	Pristina	68876	Genus	5	0.27
	Rhyacodrilus	68839	Genus	3	0.49
	Tubificinae	974289	Subfamily	4	0.21
	tubificoid Naididae without hair chaetae	–	Family	1	0.15

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Nemouridae	*Amphinemura*	102540	Genus	3	0.57
	Malenka	102567	Genus	2	0.30
	Nemouridae	102517	Family	15	0.40
	Prostoia besametsa	102585	Species	68	3.30
	Visoka cataractae	102615	Species	32	0.35
	Zapada	102591	Genus	81	1.02
	Zapada cinctipes	102594	Species	447	2.55
	Zapada columbiana	102596	Species	666	2.31
	Zapada frigida	102601	Species	1	0.16
	Zapada Oregonensis Gr.	–	Species group	153	1.07
Oreoleptidae	*Oreoleptis torrenticola*	–	Species	1	0.15
Pelecorhynchidae	*Glutops*	130915	Genus	4	0.15
Peltoperlidae	*Yoraperla brevis*	102512	Species	1	0.17
Perlidae	*Claassenia sabulosa*	102932	Species	72	0.50
	Doroneuria	103121	Genus	6	1.07
	Doroneuria theodora	103122	Species	1	0.17
	Hesperoperla pacifica	102972	Species	111	0.53
	Perlidae	102914	Family	58	0.57
Perlodidae	*Diura*	103094	Genus	24	0.52
	Isogenoides	103124	Genus	3	0.47
	Isoperla	102995	Genus	33	0.42
	Kogotus	103149	Genus	16	0.28
	Megarcys	103110	Genus	157	0.84
	Perlodidae	102994	Family	247	0.83
	Setvena	103193	Genus	1	0.15
	Skwala	103102	Genus	59	0.45
Philopotamidae	*Philopotamidae*	115257	Family	1	0.09
Phoridae	*Phoridae*	138921	Family	1	0.17
Physidae	*Physa*	76677	Genus	1	0.09
	Physella	76698	Genus	2.5	0.13
Piscicolidae	*Piscicola*	69304	Genus	1	0.18
Planariidae	*Polycelis*	54510	Genus	180	1.71
	Polycelis coronata	54512	Species	32	0.53

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Planorbidae	*Gyraulus*	76592	Genus	7	0.63
Polycentropodidae	*Polycentropodidae*	117043	Family	1	0.10
Psychodidae	*Pericoma*	125514	Genus	47	0.45
	Pericomaini	–	Family	14	0.54
	Psychodidae	125351	Family	2	0.25
Pteronarcyidae	*Pteronarcella*	102485	Genus	22	0.42
	Pteronarcella badia	102486	Species	2	0.15
	Pteronarcys	102471	Genus	6.5	0.21
	Pteronarcys californica	102473	Species	14	0.29
Rhyacophilidae	*Rhyacophila*	115097	Genus	140	0.50
	Rhyacophila Alberta Gr.	–	Species group	12	0.27
	Rhyacophila Angelita Gr.	–	Species group	12.5	0.54
	Rhyacophila arnaudi	115165	Species	1	0.33
	Rhyacophila atrata complex	–	Species complex	7	0.22
	Rhyacophila Betteni Gr.	–	Species group	420	1.61
	Rhyacophila Brunnea Gr.	–	Species group	15	0.40
	Rhyacophila Brunnea/Vemna Gr.	–	Species group	48	0.43
	Rhyacophila Hyalinata Gr.	–	Species group	152	0.81
	Rhyacophila narvae	115155	Species	3	0.24
	Rhyacophila pellisa	115114	Species	3	0.17
	Rhyacophila Rotunda Gr.	–	Genus	1	0.16
	Rhyacophila Vagrita Gr.	–	Species group	1	0.16
	Rhyacophila verrula	115125	Species	5	0.20
	Rhyacophila Vofixa Gr.	–	Species group	19	0.21
Simuliidae	*Helodon*	–	Genus	116	0.88
	Prosimuliini	126648	Tribe	2	0.61
	Prosimulium	126703	Genus	30	0.54
	Simuliidae	126640	Family	784	7.49
	Simulium	126774	Genus	2390	4.91
Sperchontidae	*Sperchon*	83006	Genus	548	1.38
	Sperchonopsis	83029	Genus	1	0.16

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Sphaeriidae	*Pisidium*	81400	Genus	2	0.20
Stygothrombiinae	*Stygothrombium*	–	Genus	1	0.16
Taeniopterygidae	*Taenionema*	102519	Genus	65	3.80
	Taeniopterygidae	102788	Family	66	0.45
Tanyderidae	*Protanyderus*	125802	Genus	7	0.19
	Tanyderidae	125799	Family	1	0.15
Tipulidae	*Antocha*	119656	Genus	35	0.49
	Antocha monticola	119660	Species	30	0.39
	Dicranota	121027	Genus	14	0.21
	Hesperoconopa	120732	Genus	2	0.17
	Hexatoma	120094	Genus	540.5	1.35
	Limnophila	120164	Genus	15	0.74
	Molophilus	120758	Genus	2	0.31
	Pedicia	121118	Genus	1	0.15
	Rhabdomastix	120968	Genus	5	0.51
	Rhabdomastix Fascigera Gr.	–	Species group	3	0.16
	Rhabdomastix Setigera Gr.	–	Species group	5	0.76
	Tipula	119037	Genus	14	0.27
	Tipulidae	118840	Family	8	0.18
Torrenticolidae	*Monatractides*	–	Genus	9	0.15
	Testudacarus	83250	Genus	2	0.15
	Torrenticola	83254	Genus	241	1.30
Uenoidae	*Neophylax*	116046	Genus	1	0.16
	Neophylax rickeri	116054	Species	0.5	0.09
	Neothremma	116388	Genus	3	0.54
	Oligophlebodes	116039	Genus	7	0.28
Unionicolidae	*Unionicolidae*	83072	Family	1	0.14
Valvatidae	*Valvata*	70346	Genus	1	0.15
Acari	*Acari*	733321	Subclass	85	0.66
Diptera	*Diptera*	118831	Order	1	0.16
Gastropoda	*Gastropoda*	69459	Class	1	0.10
Hydrozoa	*Hydrozoa*	48739	Class	1	0.15
Nemata	*Nemata*	563956	Phylum	101	0.42

Table 34 (continued). Macroinvertebrate taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Because macroinvertebrate taxa were identified by multiple taxonomists, identifications were collapsed into Operational Taxonomic Units (OTU) for comparable values.

Family or higher	Taxa (Rhithron OTU)	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Nematoda	*Nematoda*	59490	Phylum	5	0.20
Nematomorpha	*Nematomorpha*	64183	Phylum	3	0.47
Oligochaeta	*Oligochaeta*	68422	Subclass	94	1.51
Oribatida	*Oribatida*	733326	Order	7	0.21
Ostracoda	*Ostracoda*	84195	Class	1104	4.89
Plecoptera	*Plecoptera*	102467	Order	34	0.44
Trepaxonemata	*Trepaxonemata*	914204	Class	1	0.15
Trichoptera	*Trichoptera*	115095	Order	41	1.18
Turbellaria	*Turbellaria*	53964	Class	10	0.34

Diatoms

Table 35. Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Achnanthaceae	*Achnanthes*	–	Genus	7	0.29
	Achnanthes conspicua	–	Species	6	0.99
	Achnanthes curtissima	−633490	Species	2	0.28
	Achnanthes kriegeri	–	Species	6	0.50
	Achnanthes linearis	3516	Species	1	0.35
	Achnanthes minutissima v. jackii	−633608	Variety	8	0.46
	Achnanthes nodosa	–	Species	6	0.34
	Achnanthes oblongella	–	Species	1	0.16
	Achnanthes petersenii	–	Species	4	0.49
	Achnanthes rosenstockii	−633663	Species	2	0.26
	Cocconeis pediculus	–	Species	7	0.29
	Cocconeis placentula	–	Species	496	1.95
	Cocconeis placentula sensu lato	–	Species	228	1.45
	Cocconeis placentula v. lineata	–	Variety	22	0.91
	Karayevia clevei	–	Species	6	0.17
	Karayevia laterostrata	–	Species	3	0.25
	Karayevia suchlandtii	–	Species	2	0.33
	Lemnicola hungarica	–	Species	0	0.08
	Planothidium	–	Genus	19	0.23
	Planothidium apiculatum	–	Species	0	0.08
	Planothidium daui	–	Species	2	0.33
	Planothidium dubium	–	Species	14	0.26
	Planothidium frequentissimum	–	Species	124	0.67
	Planothidium haynaldii	–	Species	1	0.16
	Planothidium lanceolatum	–	Species	47	0.54
	Planothidium peragalli	–	Species	1	0.12
	Planothidium reichardtii	–	Species	2	0.33
	Planothidium rostratum	–	Species	6	0.35
	Platessa conspicua	–	Species	2	0.33
	Psammothidium	–	Genus	10	0.27
	Psammothidium bioretii	–	Species	11	0.38

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Achnanthaceae (continued)	*Psammothidium chlidanos*	–	Species	45	1.48
	Psammothidium curtissimum	–	Species	2	0.33
	Psammothidium daonense	–	Species	148	2.28
	Psammothidium subatomoides	–	Species	178	1.19
	Rossithidium nodosum	–	Species	84	1.75
	Rossithidium petersenii	–	Species	3	0.50
	Rossithidium pusillum	–	Species	57	1.05
Achnanthidiaceae	*Achnanthidium*	–	Genus	26	0.86
	Achnanthidium affine	–	Species	88	1.12
	Achnanthidium catenatum	–	Species	2	0.33
	Achnanthidium deflexum	–	Species	541	2.05
	Achnanthidium exiguum	–	Species	1	0.12
	Achnanthidium gracillimum	–	Species	886	3.06
	Achnanthidium latecephalum	–	Species	3	0.49
	Achnanthidium minutissimum	–	Species	16878	32.04
	Achnanthidium minutissimum v. jackii	–	Variety	2	0.31
	Achnanthidium neomicrocephalum	–	Species	39	0.84
	Achnanthidium pyrenaicum	–	Species	1962	5.72
	Achnanthidium rivulare	–	Species	173	0.81
	Achnanthidium rosenstockii	–	Species	1	0.17
	Achnanthidium thienemannii	–	Species	17	0.70
	Eucocconeis alpestris	–	Species	0	0.08
	Eucocconeis flexella	–	Species	28	0.23
	Eucocconeis laevis	–	Species	203	0.84
Amphipleuraceae	*Amphipleura pellucida*	–	Species	116	0.65
	Frustulia amphipleuroides	–	Species	3	0.16
	Frustulia crassinervia	–	Species	5	0.41
	Frustulia rhomboides	–	Species	0	0.08
	Frustulia saxonica	–	Species	3	0.25
	Frustulia vulgaris	–	Species	1	0.10
Aulacoseiraceae	*Aulacoseira*	–	Genus	12	0.30
	Aulacoseira alpigena	–	Species	14	0.72

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Aulacoseiraceae (continued)	*Aulacoseira canadensis*	591195	Species	1	0.15
	Aulacoseira crenulata	–	Species	13	2.16
	Aulacoseira granulata	–	Species	17	2.83
Bacillariaceae	*Denticula kuetzingii*	−634929	Species	2	0.31
	Denticula subtilis	–	Species	6	0.21
	Denticula tenuis	–	Species	49	0.47
	Grunowia tabellaria	–	Species	1	0.17
	Hantzschia	–	Genus	0	0.08
	Hantzschia amphioxys	–	Species	2	0.17
	Nitzschia	–	Genus	216	0.99
	Nitzschia acicularioides	5217	Species	2	0.31
	Nitzschia acicularis	–	Species	20	0.42
	Nitzschia acidoclinata	–	Species	33	0.62
	Nitzschia agnita	–	Species	56	1.16
	Nitzschia alpina	–	Species	12	0.46
	Nitzschia alpinobacillum	–	Species	9	0.50
	Nitzschia amphibia	–	Species	16	0.37
	Nitzschia angustata	–	Species	24	0.25
	Nitzschia angustatula	–	Species	5	0.37
	Nitzschia archibaldii	–	Species	301	2.43
	Nitzschia bacillum	–	Species	163	1.76
	Nitzschia bryophila	–	Species	36	0.55
	Nitzschia communis	–	Species	1	0.17
	Nitzschia dissipata	–	Species	472	1.71
	Nitzschia dissipata v. media	–	Variety	10	0.33
	Nitzschia draveillensis	–	Species	2	0.33
	Nitzschia flexa	–	Species	0	0.08
	Nitzschia fonticola	–	Species	119	0.68
	Nitzschia fossilis	–	Species	3	0.25
	Nitzschia frustulum	–	Species	88	0.82
	Nitzschia graciliformis	–	Species	24	0.55
	Nitzschia gracilis	–	Species	69	0.81
	Nitzschia hantzschiana	–	Species	16	0.88
	Nitzschia heufleriana	–	Species	15	0.26

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Bacillariaceae (continued)	*Nitzschia incognita*	–	Species	4	0.21
	Nitzschia inconspicua	–	Species	80	0.65
	Nitzschia intermedia	–	Species	0	0.08
	Nitzschia lacuum	–	Species	310	2.12
	Nitzschia lancettula	–	Species	13	0.41
	Nitzschia liebethruthii	–	Species	15	2.48
	Nitzschia liebetruthii	–	Species	35	1.16
	Nitzschia linearis	–	Species	51	0.39
	Nitzschia microcephala	–	Species	23	0.54
	Nitzschia palea	–	Species	90	0.57
	Nitzschia palea v. debilis	–	Variety	5	0.27
	Nitzschia paleacea	–	Species	432	2.39
	Nitzschia pellucida	–	Species	4	0.37
	Nitzschia perminuta	–	Species	414	1.46
	Nitzschia pumila	–	Species	13	0.72
	Nitzschia pura	–	Species	106	0.54
	Nitzschia pusilla	–	Species	6	0.34
	Nitzschia radicula	–	Species	2	0.28
	Nitzschia recta	–	Species	54	0.46
	Nitzschia sigmoidea	–	Species	2	0.15
	Nitzschia siliqua	–	Species	1	0.16
	Nitzschia sinuata v. tabellaria	–	Variety	1	0.17
	Nitzschia sociabilis	–	Species	3	0.16
	Nitzschia soratensis	–	Species	5	0.42
	Nitzschia subacicularis	5259	Species	4	0.28
	Nitzschia sublinearis	–	Species	16	0.27
	Nitzschia supralitorea	–	Species	6	0.50
	Nitzschia vermicularis	–	Species	13	0.28
	Simonsenia delognei	–	Species	20	0.52
	Tryblionella angustata	–	Species	5	0.19
Bacillariophyta familia incertae sedis	*Gomphonella olivacea*	–	Species	14	0.46
Berkeleyaceae	*Parlibellus protracta*	–	Species	3	0.25

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Brachysiraceae	*Brachysira brebissonii*	–	Species	1	0.17
	Brachysira microcephala	–	Species	216	1.10
	Brachysira ocalanensis	–	Species	2	0.33
	Brachysira vitrea	–	Species	40	1.32
Catenulaceae	*Amphora*	–	Genus	1	0.17
	Amphora copulata	–	Species	19	0.25
	Amphora inariensis	–	Species	10	0.22
	Amphora ovalis	–	Species	0	0.08
	Amphora pediculus	–	Species	78	0.53
Cavinulaceae	*Cavinula jaernefeltii*	–	Species	3	0.25
	Cavinula pseudoscutiformis	–	Species	12	0.18
Cymbellaceae	*Cymbella*	–	Genus	23	0.32
	Cymbella affiniformis	–	Species	75	0.86
	Cymbella affinis	–	Species	147	1.29
	Cymbella aspera	–	Species	1	0.12
	Cymbella austriaca	–	Species	4	0.67
	Cymbella blinnii	–	Species	1	0.17
	Cymbella cistula	–	Species	19	0.36
	Cymbella cleve-eulerae	–	Species	0	0.08
	Cymbella compacta	–	Species	0	0.08
	Cymbella cuspidata	–	Species	0	0.08
	Cymbella cymbiformis	–	Species	0	0.08
	Cymbella delicatula	4824	Species	7	0.37
	Cymbella ehrenbergii	–	Species	0	0.08
	Cymbella excisa	–	Species	317	1.92
	Cymbella excisiformis	–	Species	205	1.68
	Cymbella hantzschiana	4795	Species	0	0.08
	Cymbella hustedtii	–	Species	7	0.24
	Cymbella lanceolata	–	Species	1	0.11
	Cymbella maggiana	–	Species	18	0.39
	Cymbella mexicana	–	Species	0	0.08
	Cymbella naviculiformis	–	Species	3	0.16
	Cymbella neocistula	–	Species	18	0.26

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Cymbellaceae (continued)	*Cymbella neocistula v. islandica*	–	Variety	2	0.12
	Cymbella perparva	–	Species	8	0.25
	Cymbella rumrichae	–	Species	38	1.06
	Cymbella subturgidula	–	Species	35	0.55
	Cymbella vulgata	–	Species	5	0.22
	Cymbopleura	–	Genus	3	0.44
	Cymbopleura amphicephala	–	Species	4	0.25
	Cymbopleura heinii	−634896	Species	0	0.07
	Cymbopleura naviculiformis	–	Species	5	0.14
	Cymbopleura subaequalis	–	Species	4	0.63
	Cymbopleura subcuspidata	–	Species	0	0.08
	Delicata	–	Genus	0	0.08
	Delicata alpestris	–	Species	4	0.33
	Delicata delicatula	–	Species	12	0.30
	Delicata montana	–	Species	1	0.10
	Didymosphenia geminata	–	Species	51	0.24
	Encyonema	–	Genus	10	0.27
	Encyonema auerswaldii	–	Species	5	0.41
	Encyonema cespitosum	–	Species	16	0.40
	Encyonema elginense	−635153	Species	2	0.31
	Encyonema fogedii	–	Species	122	2.90
	Encyonema gaeumannii	–	Species	2	0.28
	Encyonema latens	–	Species	21	0.87
	Encyonema minutum	–	Species	673	2.02
	Encyonema muelleri	591269	Species	10	0.26
	Encyonema neogracile	–	Species	19	0.51
	Encyonema neomesianum	–	Species	4	0.25
	Encyonema obscurum	–	Species	1	0.16
	Encyonema perminutum	–	Species	2	0.33
	Encyonema prostratum	–	Species	4	0.27
	Encyonema reichardtii	–	Species	235	1.69
	Encyonema silesiacum	–	Species	1416	2.87
	Encyonema ventricosum	–	Species	75	0.65

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Cymbellaceae (continued)	*Encyonema vulgare*	–	Species	3	0.42
	Encyonema yellowstonianum	–	Species	0	0.08
	Encyonopsis	–	Genus	6	0.36
	Encyonopsis alpina	–	Species	15	1.24
	Encyonopsis cesatiformis	–	Species	18	0.50
	Encyonopsis cesatii	–	Species	69	0.57
	Encyonopsis czarneckii	–	Species	3	0.16
	Encyonopsis descripta	–	Species	7	0.58
	Encyonopsis falaisensis	–	Species	0	0.08
	Encyonopsis hustedtii	–	Species	3	0.50
	Encyonopsis krammeri	–	Species	10	0.84
	Encyonopsis microcephala	–	Species	202	1.52
	Encyonopsis minuta	–	Species	4	0.68
	Encyonopsis montana	–	Species	7	0.22
	Encyonopsis perborealis	–	Species	365	5.75
	Encyonopsis stafsholtii	–	Species	5	0.41
	Encyonopsis subminuta	–	Species	1126	3.51
	Navicymbula pusilla	–	Species	2	0.33
	Placoneis	–	Genus	1	0.16
	Placoneis elginensis	–	Species	1	0.12
	Placoneis explanata	–	Species	5	0.24
Diadesmidaceae	*Diadesmis perpusilla*	–	Species	7	0.53
	Humidophila schmassmannii	–	Species	6	0.50
	Luticola	–	Genus	0	0.08
	Luticola mutica	–	Species	0	0.08
Diatomaceae	*Distrionella incognita*	–	Species	10	0.47
Diploneidaceae	*Diploneis alpina*	−635020	Species	0	0.08
	Diploneis hinziae	–	Species	1	0.17
	Diploneis krammeri	–	Species	0	0.08
	Diploneis oblongella	–	Species	2	0.21
	Diploneis ovalis	–	Species	6	0.23

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Eunotiaceae	*Eunotia*	–	Genus	14	0.43
	Eunotia arcus	–	Species	2	0.21
	Eunotia bilunaris	–	Species	4	0.29
	Eunotia bilunaris v. mucophila	–	Variety	3	0.49
	Eunotia diadema	–	Species	3	0.50
	Eunotia diodon	–	Species	11	1.82
	Eunotia glacialis	–	Species	2	0.33
	Eunotia implicata	–	Species	2	0.33
	Eunotia intermedia	–	Species	0	0.08
	Eunotia minor	–	Species	29	0.94
	Eunotia monodon	3405	Species	1	0.35
	Eunotia paludosa	–	Species	1	0.17
	Eunotia praerupta	–	Species	2	0.13
	Eunotia subarcuatoides	–	Species	5	0.42
Fragilariaceae	*Diatoma anceps*	–	Species	150	2.47
	Diatoma hiemalis	–	Species	13	0.23
	Diatoma hyemale	–	Species	1	0.12
	Diatoma hyemalis	–	Species	0	0.08
	Diatoma mesodon	–	Species	117	0.65
	Diatoma moniliformis	–	Species	298	1.80
	Diatoma tenuis	–	Species	257	1.23
	Diatoma vulgaris	–	Species	0	0.08
	Fragilaria	–	Genus	43	0.47
	Fragilaria capucina	–	Species	254	1.23
	Fragilaria capucina v. gracilis	–	Variety	149	1.12
	Fragilaria capucina v. mesolepta	–	Variety	6	0.34
	Fragilaria capucina v. perminuta	–	Variety	4	0.68
	Fragilaria crotonensis	–	Species	30	0.41
	Fragilaria gracilis	–	Species	37	0.77
	Fragilaria mesolepta	–	Species	1	0.16
	Fragilaria microvaucheriae	–	Species	1	0.16
	Fragilaria nanana	–	Species	52	0.67

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Fragilariaceae (continued)	*Fragilaria pectinalis*	–	Species	27	0.56
	Fragilaria sepes	–	Species	54	0.78
	Fragilaria tenera	–	Species	70	1.18
	Fragilaria vaucheriae	–	Species	534	1.91
	Fragilaria vaucheriae morphotype A GLAC LLB	2942	Species	179	1.69
	Fragilaria vaucheriae morphotype B GLAC LLB	2942	Species	69	1.13
	Fragilaria vaucheriae v. capitellata	2945	Variety	15	0.68
	Hannaea arcus	–	Species	2028	5.16
	Meridion anceps	–	Species	16	0.89
	Meridion circulare	–	Species	93	0.53
	Meridion circulare v. constrictum	–	Variety	23	0.97
	Odontidium mesodon	–	Species	7	0.39
	Opephora olsenii	–	Species	2	0.33
	Pseudostaurosira	–	Genus	3	0.50
	Pseudostaurosira brevistriata	–	Species	146	0.92
	Pseudostaurosira elliptica	–	Species	98	2.51
	Pseudostaurosira parasitica	–	Species	2	0.33
	Pseudostaurosira pseudoconstruens	–	Species	50	1.64
	Pseudostaurosira sp. 404201 GLAC LLB	590851	Genus	10	1.57
	Staurosira	–	Genus	4	0.29
	Staurosira construens	–	Species	459	2.43
	Staurosira construens v. binodis	–	Variety	12	0.49
	Staurosira construens v. pumila	–	Variety	3	0.25
	Staurosira construens v. venter	–	Variety	1669	6.09
	Staurosira elliptica	–	Species	4	0.68
	Staurosira venter	–	Species	156	1.87
	Staurosirella	–	Genus	4	0.66

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Fragilariaceae (continued)	*Staurosirella dubia*	–	Species	2	0.34
	Staurosirella lapponica	–	Species	10	0.52
	Staurosirella leptostauron	–	Species	86	0.43
	Staurosirella leptostauron v. dubia	–	Variety	102	2.80
	Staurosirella martyi	–	Species	205	2.98
	Staurosirella ovata	–	Species	149	4.31
	Staurosirella pinnata	–	Species	1982	5.58
	Staurosirella rhomboides	–	Species	6	1.02
	Synedra acus	–	Species	153	1.12
	Synedra biceps	–	Species	11	0.85
	Synedra delicatissima	–	Species	2	0.15
	Synedra mazamaensis	–	Species	7	0.37
	Synedra rumpens	–	Species	127	1.06
	Synedra ulna	–	Species	158	0.87
	Synedra ulna v. chaseana	–	Variety	1	0.12
	Ulnaria acus	–	Species	35	0.43
	Ulnaria ulna	–	Species	13	0.26
Gomphonema-taceae	*Gomphoneis*	–	Genus	2	0.33
	Gomphoneis eriense	–	Species	24	0.55
	Gomphoneis eriense v. variabilis	–	Variety	6	1.00
	Gomphoneis geitleri	–	Species	337	2.24
	Gomphoneis herculeana v. abundans	–	Variety	0	0.08
	Gomphoneis minuta	–	Species	4	0.20
	Gomphoneis olivaceoides	−636026	Species	439	5.50
	Gomphoneis olivaceum	–	Species	75	1.54
	Gomphoneis pseudookunoi	–	Species	75	12.50
	Gomphoneis septa	–	Species	12	0.38
	Gomphonema	–	Genus	1107	3.46
	Gomphonema acuminatum	–	Species	0	0.08
	Gomphonema aff. longilineare GLAC LLB	−636027	Species	69	0.99
	Gomphonema affine	4956	Species	2	0.28

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Gomphonema-taceae (continued)	*Gomphonema americobtusatum*	–	Species	33	0.84
	Gomphonema angustatum	–	Species	480	3.47
	Gomphonema angustum	–	Species	7	1.17
	Gomphonema bipunctatum	−636065	Species	2	0.28
	Gomphonema calcifugum	–	Species	2	0.33
	Gomphonema cf. minutum GLAC LLB	4911	Species	8	0.62
	Gomphonema cf. pumilum GLAC LLB	4911	Species	14	0.64
	Gomphonema citera	–	Species	332	4.36
	Gomphonema clavatulum	−636084	Species	0	0.07
	Gomphonema clavatum	–	Species	8	1.30
	Gomphonema cymbelliclinum	–	Species	98	2.05
	Gomphonema drutelingense	–	Species	4	0.15
	Gomphonema exilissimum	–	Species	1	0.16
	Gomphonema gracile	–	Species	9	0.39
	Gomphonema innocens	–	Species	4	0.67
	Gomphonema insigne	–	Species	1	0.17
	Gomphonema instabilis	–	Species	1	0.35
	Gomphonema intricatum	4949	Species	0	0.08
	Gomphonema kobayasii	–	Species	254	1.62
	Gomphonema longilineare	–	Species	36	1.99
	Gomphonema mexicanum	–	Species	0	0.08
	Gomphonema micropus	–	Species	414	2.39
	Gomphonema minusculum	–	Species	294	1.71
	Gomphonema minutum	–	Species	202	1.40
	Gomphonema olivaceoides	–	Species	376	4.50
	Gomphonema olivaceoides v. densestriata	–	Variety	6	0.50
	Gomphonema olivaceum	–	Species	117	1.49
	Gomphonema pala	–	Species	0	0.08
	Gomphonema parvulum	–	Species	44	0.74
	Gomphonema pseudotenellum	591244	Species	6	1.02
	Gomphonema pumilum	–	Species	722	4.49

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Gomphonemataceae (continued)	*Gomphonema pumilum v. elegans*	–	Variety	50	1.66
	Gomphonema pumilum v. rigidum	–	Variety	92	3.06
	Gomphonema pygmaeum	4911	Species	65	4.77
	Gomphonema quadripunctatum	–	Species	104	3.46
	Gomphonema rhombicum	–	Species	0	0.08
	Gomphonema sp. 408705 GLAC LLB	4911	Genus	2	0.26
	Gomphonema subclavatum	–	Species	68	2.79
	Gomphonema truncatum	–	Species	3	0.16
	Gomphonema utae	–	Species	15	1.23
	Gomphonema ventricosum	–	Species	2	0.20
	Gomphonema vibrio	–	Species	23	1.93
	Gomphosinica geitleri	–	Species	51	0.94
	Gomphosphenia	–	Genus	4	0.28
	Reimeria sinuata	–	Species	1265	2.54
	Reimeria uniseriata	–	Species	1	0.11
Mastogloiaceae	*Mastogloia grevillei*	–	Species	0	0.08
Melosiraceae	*Melosira varians*	–	Species	2	0.29
Naviculaceae	*Adlafia bryophila*	–	Species	59	1.18
	Adlafia langebertalotii	–	Species	107	8.90
	Adlafia minuscula	–	Species	186	1.18
	Adlafia suchlandtii	–	Species	138	4.58
	Chamaepinnularia soehrensis	–	Species	5	0.42
	Eolimna minima	–	Species	218	1.10
	Geissleria acceptata	–	Species	13	0.37
	Geissleria decussis	–	Species	14	0.44
	Geissleria similis	–	Species	2	0.33
	Genkalia boreoalpina	–	Species	2	0.33
	Hippodonta capitata	–	Species	0	0.08
	Hippodonta coxiae	–	Species	1	0.16

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Naviculaceae (continued)	*Hippodonta lueneburgensis*	–	Species	0	0.08
	Mayamaea agrestis	–	Species	4	0.66
	Mayamaea atomus	–	Species	48	0.56
	Mayamaea permitis	–	Species	1	0.17
	Navicula	–	Genus	73	0.53
	Navicula absoluta	–	Species	1	0.16
	Navicula aitchelbee	–	Species	3	0.29
	Navicula angusta	–	Species	6	0.34
	Navicula antonii	–	Species	46	0.53
	Navicula arctotenelloides	–	Species	14	0.28
	Navicula arvensis	–	Species	9	0.37
	Navicula associata	−636860	Species	2	0.19
	Navicula aurora	–	Species	0	0.08
	Navicula capitatoradiata	–	Species	30	0.27
	Navicula cari	–	Species	10	0.82
	Navicula caroliniae	–	Species	1	0.10
	Navicula catalanogermanica	−636947	Species	2	0.32
	Navicula caterva	–	Species	134	2.02
	Navicula constans	3833	Species	0	0.07
	Navicula cryptocephala	–	Species	99	0.51
	Navicula cryptofallax	–	Species	50	1.22
	Navicula cryptotenella	–	Species	247	0.69
	Navicula cryptotenelloides	–	Species	2	0.21
	Navicula densilineolata	–	Species	4	0.33
	Navicula detenta	–	Species	2	0.33
	Navicula disjuncta	–	Species	1	0.16
	Navicula erifuga	–	Species	2	0.33
	Navicula exilis	–	Species	2	0.34
	Navicula germainii	–	Species	4	0.67
	Navicula glomus	−637181	Species	2	0.31
	Navicula goersii	–	Species	2	0.33
	Navicula gregaria	–	Species	9	0.27
	Navicula helensis	4026	Species	2	0.29
	Navicula hintzii	–	Species	1	0.35

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Naviculaceae (continued)	*Navicula hustedtii*	–	Species	4	0.37
	Navicula kotschyi	–	Species	6	0.33
	Navicula libonensis	–	Species	2	0.33
	Navicula lundii	–	Species	2	0.33
	Navicula menisculus	–	Species	6	0.25
	Navicula metareichardtiana	–	Species	6	1.00
	Navicula modica	193791	Species	0	0.08
	Navicula moskalii	–	Species	2	0.20
	Navicula notha	–	Species	3	0.49
	Navicula oligotraphenta	−637453	Species	3	0.43
	Navicula oppugnata	3913	Species	2	0.30
	Navicula parablis	–	Species	2	0.33
	Navicula peregrina	–	Species	2	0.12
	Navicula phyllepta	–	Species	0	0.08
	Navicula pseudosilicula f. olympica	–	Species	0	0.08
	Navicula pseudotenelloides	–	Species	2	0.32
	Navicula radiosa	–	Species	63	0.28
	Navicula radiosafallax	–	Species	0	0.08
	Navicula recens	–	Species	9	0.37
	Navicula reichardtiana	–	Species	145	1.77
	Navicula reinhardtii	–	Species	0	0.08
	Navicula rhynchocephala	–	Species	0	0.08
	Navicula rhynchotella	–	Species	0	0.08
	Navicula rostellata	–	Species	2	0.33
	Navicula schmassmannii	–	Species	4	0.37
	Navicula seibigiana	–	Species	6	0.22
	Navicula staffordiae	–	Species	4	0.66
	Navicula stroemii	3982	Species	7	0.37
	Navicula subconcentrica	–	Species	2	0.20
	Navicula subminuscula	–	Species	0	0.08
	Navicula subtilissima	–	Species	1	0.17
	Navicula supleeorum	–	Species	1	0.17
	Navicula tenelloides	–	Species	10	0.33

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Naviculaceae (continued)	*Navicula tripunctata*	–	Species	10	0.18
	Navicula trivialis	–	Species	11	0.22
	Navicula veneta	–	Species	7	0.39
	Navicula ventralis	–	Species	8	0.35
	Navicula viridula	–	Species	5	0.17
	Navicula viridulacalcis	–	Species	3	0.23
	Navicula vitabunda	–	Species	3	0.24
	Navicula vulpina	–	Species	4	0.18
	Navicula weberi	–	Species	0	0.08
	Navicula wiesneri	–	Species	15	0.83
	Nupela lapidosa	–	Species	7	1.16
	Nupela subrostrata	–	Species	1	0.17
Neidiaceae	*Neidiopsis levanderi*	–	Species	1	0.17
	Neidium affine	–	Species	1	0.12
	Neidium alpinum	–	Species	0	0.08
	Neidium ampliatum	–	Species	2	0.10
	Neidium binodeformis	–	Species	0	0.07
	Neidium bisulcatum	–	Species	1	0.12
	Neidium dubium	–	Species	2	0.15
	Neidium productum	3310	Species	2	0.28
	Undetermined Pennate	–	Subclass	72	1.98
Pinnulariaceae	*Caloneis*	–	Genus	6	0.20
	Caloneis bacillum	–	Species	98	0.67
	Caloneis lancettula	–	Species	13	2.16
	Caloneis schumanniana	–	Species	4	0.19
	Caloneis silicula	–	Species	14	0.28
	Caloneis tenuis	–	Species	14	0.38
	Diatomella balfouriana	–	Species	28	0.79
	Pinnularia	–	Genus	13	0.19
	Pinnularia borealis	–	Species	3	0.13
	Pinnularia divergens	4473	Species	0	0.08
	Pinnularia gibba	–	Species	2	0.33
	Pinnularia interrupta	–	Species	0	0.08
	Pinnularia latarea	–	Species	0	0.08

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Pinnulariaceae (continued)	*Pinnularia microstauron*	–	Species	15	0.39
	Pinnularia nodosa v. percapitata	−638764	Variety	0	0.08
	Pinnularia pulchra	–	Species	0	0.08
	Pinnularia rhombarea	−638837	Species	2	0.71
	Pinnularia septentrionalis	–	Species	0	0.08
	Pinnularia subcapitata	–	Species	0	0.08
	Pinnularia viridis	–	Species	0	0.08
Pleurosig-mataceae	*Gyrosigma acuminatum*	–	Species	0	0.08
	Gyrosigma nodiferum	–	Species	1	0.16
	Gyrosigma spencerii	–	Species	0	0.08
Rhoicospheni-aceae	*Rhoicosphenia abbreviata*	–	Species	28	0.39
Rhopalodiaceae	*Epithemia adnata*	–	Species	54	0.55
	Epithemia argus	5006	Species	11	1.71
	Epithemia gibba	–	Species	12	1.99
	Epithemia sorex	–	Species	145	1.09
	Epithemia turgida	–	Species	67	0.43
	Rhopalodia gibba	–	Species	65	0.47
	Rhopalodia gibberula	–	Species	0	0.08
Sellaphoraceae	*Fallacia pygmaea*	–	Species	1	0.12
	Fallacia subhamulata	591218	Species	0	0.08
	Fallacia tenera	–	Species	4	0.66
	Pseudofallacia tenera	–	Species	1	0.16
	Sellaphora	–	Genus	24	0.66
	Sellaphora atomoides	–	Species	23	0.54
	Sellaphora bacilloides	–	Species	1	0.16
	Sellaphora bacillum	–	Species	7	0.25
	Sellaphora difficillima	–	Species	1	0.16
	Sellaphora disjuncta	–	Species	4	0.66
	Sellaphora laevissima	–	Species	10	0.24
	Sellaphora medioconvexa	–	Species	2	0.33
	Sellaphora mutata	591397	Species	0	0.08
	Sellaphora nigri	–	Species	97	1.14

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Sellaphoraceae (continued)	*Sellaphora pupula*	–	Species	35	0.25
	Sellaphora seminulum	–	Species	7	0.29
Stauroneidaceae	*Craticula ambigua*	–	Species	0	0.08
	Craticula cuspidata	–	Species	0	0.08
	Craticula molestiformis	–	Species	1	0.16
	Stauroneis	–	Genus	0	0.08
	Stauroneis anceps	–	Species	3	0.12
	Stauroneis dilatata	4316	Species	1	0.16
	Stauroneis gracilis	–	Species	3	0.15
	Stauroneis italica	–	Species	0	0.08
	Stauroneis kriegeri	–	Species	4	0.33
	Stauroneis lauenburgiana	–	Species	1	0.16
	Stauroneis pergracilis	–	Species	0	0.18
	Stauroneis phoenicenteron	–	Species	1	0.11
	Stauroneis smithii	–	Species	6	0.21
Stephanodis-caceae	*Cyclotella*	–	Genus	6	0.54
	Cyclotella antiqua	–	Species	0	0.08
	Cyclotella bodanica v. lemanica	–	Variety	0	0.08
	Cyclotella comensis	2464	Species	4	0.52
	Cyclotella comta	–	Species	1	0.16
	Cyclotella cyclopuncta	–	Species	11	0.90
	Cyclotella distinguenda	–	Species	5	0.83
	Cyclotella michiganiana	2471	Species	1	0.15
	Lindavia delicatula	–	Species	7	0.58
Surirellaceae	*Cymatopleura elliptica*	–	Species	0	0.08
	Cymatopleura solea	–	Species	1	0.09
	Surirella	–	Genus	0	0.08
	Surirella amphioxys	–	Species	0	0.08
	Surirella angusta	–	Species	14	0.31
	Surirella librile	–	Species	2	0.33
	Surirella linearis	–	Species	5	0.17
	Surirella splendida	–	Species	0	0.08

Table 35 (continued). Diatom taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)	Mean relative abundance
Tabellariaceae	*Tabellaria fenestrata*	–	Species	6	0.34
	Tabellaria flocculosa	–	Species	69	0.45
Thalassiosiraceae	*Discostella pseudostelligera*	–	Species	2	0.33

Soft Bodied Algae

Table 36. Soft bodied algae taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Taxa identified at a level above family were included in the Family column.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)
Batrachospermaceae	*Batrachospermum*	11665	Genus	28
Borziaceae	*Borzia*	1097	Genus	11
	Komvophoron	–	Genus	18
Chaetophoraceae	*Gongrosira*	–	Genus	6
	Stigeoclonium	–	Genus	2045
Chamaesiphonaceae	*Chamaesiphon*	–	Genus	361
	Geitleribactron	–	Genus	252
Chantransiacea	*Audouinella*	–	Genus	32
	Audouinella hermannii	183017	Species	355
Chlamydomonadaceae	*Chlamydomonas*	–	Genus	2
Chroococcaceae	*Chroococcus*	654	Genus	34
	Gloeocapsopsis	–	Genus	318
Coleochaetaceae	*Coleochaete*	–	Genus	1
Cryptomonadaceae	*Cryptomonas*	–	Genus	5
Desmidiaceae	*Closterium*	–	Genus	22
	Cosmarium	–	Genus	32
	Euastrum	–	Genus	1
	Pleurotaenium	–	Genus	1
	Staurastrum	–	Genus	11
Dinobryaceae	*Dinobryon*	–	Genus	18
Euglenaceae	*Phacus*	9766	Genus	1
	Trachelomonas	–	Genus	3
Gayraliaceae	*Hazenia mirabilis*	−643496	Species	35
Homoeotrichaceae	*Heteroleibleinia*	–	Genus	890
Hydrococcaceae	*Hydrococcus*	–	Genus	143
Hydrodictyaceae	*Pediastrum*	–	Genus	3
Hydruraceae	*Hydrurus*	–	Genus	108
	Hydrurus foetidus	1873	Species	1210
Hyellaceae	*Pleurocapsa*	1435	Genus	12
Lemaneaceae	*Chantransia*	–	Genus	8
Lithodermataceae	*Heribaudiella fluviatilis*	–	Species	769

Table 36 (continued). Soft bodied algae taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Taxa identified at a level above family were included in the Family column.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)
Merismopediaceae	*Aphanocapsa*	–	Genus	7
	Gomphosphaeria	714	Genus	24
	Merismopedia	–	Genus	96
	Synechocystis	799	Genus	102
Microchaetaceae	*Tolypothrix*	–	Genus	42
Microcystaceae	*Eucapsis*	812	Genus	8
Microsporaceae	*Microspora*	–	Genus	2
Nostocaceae	*Anabaena*	–	Genus	227
	Cylindrospermum	–	Genus	2
	Nostoc	–	Genus	19
Oedogoniaceae	*Bulbochaete*	9115	Genus	12
	Oedogonium	–	Genus	24
Oocystaceae	*Ankistrodesmus*	–	Genus	49
	Monoraphidium	–	Genus	19
	Oocystis	5827	Genus	8
	Selenastrum	6249	Genus	3
Oscillatoriaceae	*Lyngbya*	–	Genus	17
	Oscillatoria	–	Genus	51
Phormidiaceae	*Phormidium*	–	Genus	1181
	Pseudophormidium	189422	Genus	21
Pseudanabaenaceae	*Geitlerinema*	–	Genus	4
	Homoeothrix	–	Genus	1884
	Homoeothrix janthina	1310	Species	4927
	Jaaginema	–	Genus	181
	Leptolyngbya	–	Genus	1228
	Limnothrix redekei	–	Species	22
	Pseudanabaena	–	Genus	220
Rivulariaceae	*Amphithrix*	1285	Genus	19
	Calothrix	–	Genus	997
	Dichothrix	1230	Genus	99
	Rivularia	–	Genus	7
Scenedesmaceae	*Scenedesmus*	–	Genus	88
	Tetrastrum	6260	Genus	8

Table 36 (continued). Soft bodied algae taxa list and relative abundances from stream ecological integrity monitoring at sentinel sites in Glacier National Park, 2007–2019. Total counts are followed by mean relative abundances. Taxa identified at a level above family were included in the Family column.

Family or higher	Taxon	Taxonomic serial number	Level of identification	Total count (2007–2019)
Schizotrichaceae	*Schizothrix*	667866	Genus	4
Synechococcaceae	*Gloeothece*	703	Genus	187
Tribonemataceae	*Tribonema*	–	Genus	18
Ulotrichaceae	*Klebsormidium*	–	Genus	97
	Ulothrix	–	Genus	26
	Ulothrix zonata	6431	Species	118
Zygnemataceae	*Mougeotia*	–	Genus	110
	Spirogyra	–	Genus	21
	Zygnema	–	Genus	61

The Department of the Interior protects and manages the nation’s natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

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