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Central Alaska Network Flowing Waters Monitoring Program

2009 Annual Report

Natural Resource Technical Report NPS/CAKN/NRTR—2011/454

ON THE COVER
Ecologist Greta Burkart at Tanada Creek, Wrangell-St. Elias National Park and Preserve
Photograph by: Trey Simmons

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All manuscripts in the series receive the appropriate level of peer review to ensure that the information is scientifically credible, technically accurate, appropriately written for the intended audience, and designed and published in a professional manner. This report received informal peer review by subject-matter experts who were not directly involved in the collection, analysis, or reporting of the data. Data in this report were collected and analyzed using methods based on established, peer-reviewed protocols and were analyzed and interpreted within the guidelines of the protocols.

This report is available from the Central Alaska Network website (http://science.nature.nps.gov/im/units/cakn/reportpubs.cfm) and the Natural Resource Publications Management website (http://www.nature.nps.gov/publications/nrpm/).

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Abstract

The 2009 field season marked the fourth year of development of the flowing waters portion of the Central Alaska Network (CAKN) Inventory and Monitoring Program, also known as the Vital Signs Program. Data collection occurred in both Wrangell-St. Elias National Park and Preserve (WRST) and in Denali National Park and Preserve (DENA). The purposes of the study were to 1) continue to refine field protocols and logistics related to the collection of relevant data in DENA and WRST streams and rivers; 2) test the feasibility of implementing a multi-panel sampling design in WRST; and 3) implement the long-term flowing water monitoring program. The data collected included biological (benthic macroinvertebrate and diatom samples, fish identity and size, riparian vegetation type), physical (channel geometry, substrate, *etc*.) and chemical (collected *in situ* as well as in water samples for later analysis) information. Data were collected during a total of 43 site visits across the two park units; 32 unique sites were sampled. Continuously recording temperature data loggers were deployed at 18 sites, and two continuously recording pressure transducers were deployed at one site, with one remaining in place over the winter. In DENA, another seven previously unreported macroinvertebrate taxa were documented, bringing the three-year total to 38 taxa. Network-wide, the program has now documented 143 unique macroinvertebrate taxa and 396 diatom species. The third year of data collected from streams involved in the Long-Term Ecological Monitoring program in DENA confirmed earlier indications that there are incompatibilities between data collected using LTEM methodologies and those collected using CAKN methodologies. Preliminary analyses were conducted on the year-to-year variability of water chemistry and macroinvertebrate data collected since 2006. A substantial portion of the temporal variation in water chemistry, at least for some streams and some constituents, appeared to be attributable to sampling during high- and/or low-flow events. Year-to-year variation in macroinvertebrate composition at most streams appears to be moderate by comparison, and nearly two thirds of the apparent variation is likely due to sampling error. Invertebrate and environmental data collected from 2006-2009 were used to construct preliminary RIVPACS predictive biological assessment models for network streams. Initial results were promising, with precision and bias of the models comparable to that seen in other large-scale models.

Acknowledgements

I would like to thank the staff of Wrangell-St. Elias National Park and Preserve, in particular Eric Veach, Thelma Schrank and everyone at the Gulkana hangar for their invaluable assistance with field logistics, and for many discussions about the park and the monitoring program. I'd also like to thank the staff of Denali National Park and Preserve, in particular Lucy Tyrell, for lots of cheerful assistance. Thanks also to Pam Sousanes and Amy Larsen for sharing helicopter time, and to Heidi Kristenson for field assistance. Finally, I'd like to thank Maggie MacCluskie for her unflagging support, intellectual, financial and moral, as I struggled through my fourth field season. And a very special thanks to Greta Burkart for invaluable contributions to the scientific development of the CAKN stream monitoring program, and for extensive help in the field.

Introduction

This study is part of the National Park Service Vital Signs (Inventory and Monitoring) Program for the Central Alaska Network (CAKN; MacCluskie and Oakley 2005). Climate change and other anthropogenic impacts can be expected to have a dramatic effect on freshwater ecosystems in Alaskan National Parks; the streams and rivers portion of the Vital Signs program has been designed to detect trends in the status of important components of lotic ecosystems. These include hydrologic regime, geomorphology, temperature, water quality and the distribution and abundance of freshwater fish, benthic macroinvertebrate and diatom species. Fundamentally, the goal is to develop a logistically feasible, repeatable and scientifically robust monitoring program that will detect change in these systems. To the extent possible, we are incorporating indicators, data and methods developed as part of the Denali National Park and Preserve (DENA) Long Term Ecological Monitoring (LTEM) program, as well as utilizing other relevant data collected in network parks for a variety of purposes.

In 2009, the purposes of the program were to continue to evaluate and refine the existing field methods, maintain the continuity of existing data streams by sampling sites along the DENA park road that were part of the LTEM program, and collect data from a variety of sites in Wrangell-St. Elias National Park and Preserve (WRST) to continue characterization of the natural variability of stream ecosystems in that park unit. In addition, a major focus in 2009 was to evaluate the feasibility of formally implementing a multi-tiered sampling design for the network. The design includes annual sampling at a small number of accessible “sentinel” sites, along with long rotation sampling at a large number of “synoptic” sites that were selected using the generalized random tessellation stratified (GRTS) algorithm, which generates a spatially-balanced, probabilistic sample (Overton and Stehman 1993, Stevens and Olsen 2004). Frequent sampling at sentinel sites will provide sensitivity to trends in various metrics, whereas parkwide inference will be established for each park unit using the synoptic GRTS sites. The ultimate goal for WRST is to sample 10 synoptic sites per year, on an approximately 10-year return interval, for a total of 100 sites. For DENA, the goal is to visit six sites per year on an approximately 10-year return interval, for a total of 60 synoptic sites; the return intervals and sample panels are still being evaluated for YUCH.

In 2008, the feasibility of using GRTS to select a probabilistic sample of synoptic sites was tested in WRST. After editing the National Hydrography Dataset to develop a workable base dataset, a list of 400 potential sampling sites was generated using the GRTS algorithm. This list of sites was stratified into “accessible” and “inaccessible” strata using a relatively simple cost surface approach. Unequal weighting was applied to the list to overrepresent wadeable stream segments using Strahler stream order as a surrogate for stream size. Remotely sensed data were used to evaluate each of the 400 sites to determine whether it was part of the target population (*e.g.,* was it actually a stream), and whether it was likely to be sampleable. Of the 305 sites that remained on the list after this step, 116 were evaluated in the field, primarily by helicopter overflight. The results of this evaluation were used to select potential sampling sites for the 2009 field season.

The logistical challenges of working in remote Alaskan parklands forced the CAKN to reconsider the strict application of the GRTS sampling methodology. In traditional GRTS sampling, sites are sampled in numerical order, moving down a list. If a particular site cannot be sampled, it is simply eliminated from the target population, and the next site on the list is sampled. However, this approach is unrealistic for the Central Alaska Network, where consecutive sites may be hundreds of miles apart, and require helicopter access as well. For example, the first GRTS site on the list for WRST, WRST-GRTS-001, is a tributary to Malaspina Lake, or possibly a side channel of the tributary. Accessing this site would require basing a helicopter or floatplane out of Yakutat. Should that site not be sampleable (or if there is no nearby landing site for a helicopter or floatplane), the next site to be sampled would be WRST-GRTS-002, which turns out to be Jack Creek, some 200 miles northwest. Although this site is accessible by road, the next site on the list is a tributary to the Copper River nearly 100 miles to the west that would require helicopter access. Similarly, the fourth site is over 100 miles east of the third and would also require helicopter access. Given the resources available (generally four or five days of helicopter use per year, almost always shared with other projects and based in a single location), this sort of strictly sequential sampling is not feasible. Accordingly, the CAKN has adopted a modified approach that maximizes the cost-effectiveness of data collection by de-emphasizing the ordering of the sites and instead emphasizing logistics. In practice, this means that in any given year the sites visited will be in geographic proximity to one another, and that certain areas of the park (*e.g*. coastal, far northeast) are probably going to be underrepresented in the final site list, as it is more difficult to arrange shared helicopter time in those areas.

In 2009, the CAKN was able to sample 12 sites from the GRTS list in Wrangell-St. Elias. One of these, WRST-GRTS-130 (Jack Creek), coincided with a road-accessible sentinel site that had been sampled since 2006; the other 11 were reached using a helicopter. In addition, seven sentinel sites in WRST were sampled, along with 14 sites in DENA. This suggests that the two-tiered survey design outlined above is feasible, although the effect of adding GRTS sites in DENA and YUCH is difficult to evaluate.

The last focus of the 2009 field season was to evaluate the feasibility of collecting quasi-replicated diatom and macroinvertebrate samples to potentially allow the development of occupancy models, which explicitly incorporate detectability (sampling error) to generate more precise estimates of species presence. Ideally eight true replicate samples would be collected at each site (corresponding to the 8 subsamples that are collected for reach composite samples), with 400 (diatoms) or 500 (invertebrates) organisms identified from each replicate. However, budgetary constraints forced the program to cut back to four replicate samples, with the total number of organisms identified remaining the same (*i.e*., for invertebrates, 125 organisms were identified per replicate). Several years of replicate samples will be required before the utility of occupancy models can be evaluated.

Invertebrate and environmental data collected from 2009 were combined with data from earlier years to develop and evaluate RIVPACS-type biological assessment models. These methods use natural environmental gradients to predict the invertebrate community that would be expected at a given site in the absence of anthropogenic stress. Deviations from the expected community composition serve as a metric of ecosystem impairment. As such, they have the potential to allow the CAKN to evaluate contemporaneous ecological integrity in network streams as well as detect changes in otherwise pristine streams due to the effects of remote stressors such as climate change.

Methods

Study Area

Data were collected from a total of 57 site visits at 33 unique sites in 2009. In DENA there were 27 site visits to 14 unique sites, and in WRST there were 30 visits to 19 unique sites (Table 1). The full suite of data was not collected during all visits, however. The locations of the sites sampled are shown in Figures 1 and 2. All of the DENA sites were located along or near the park road. These sites included 10 sites that have been sampled since 1994 as part of the LTEM program. Thirteen of the sites had been sampled by the CAKN in either 2007 or 2008; Stony Creek was sampled for the first time in 2009. In WRST, 10 sites along the road system were sampled, seven of which had been sampled in previous years. The new road sites were all from the GRTS list, and included Tanada Creek, Caribou Creek and a small tributary of Jack Creek (the latter two sites were both dry at the time of the sampling visit, although flowing earlier in the year). Nine remote sites, all of them from the GRTS list, were also sampled by helicopter.

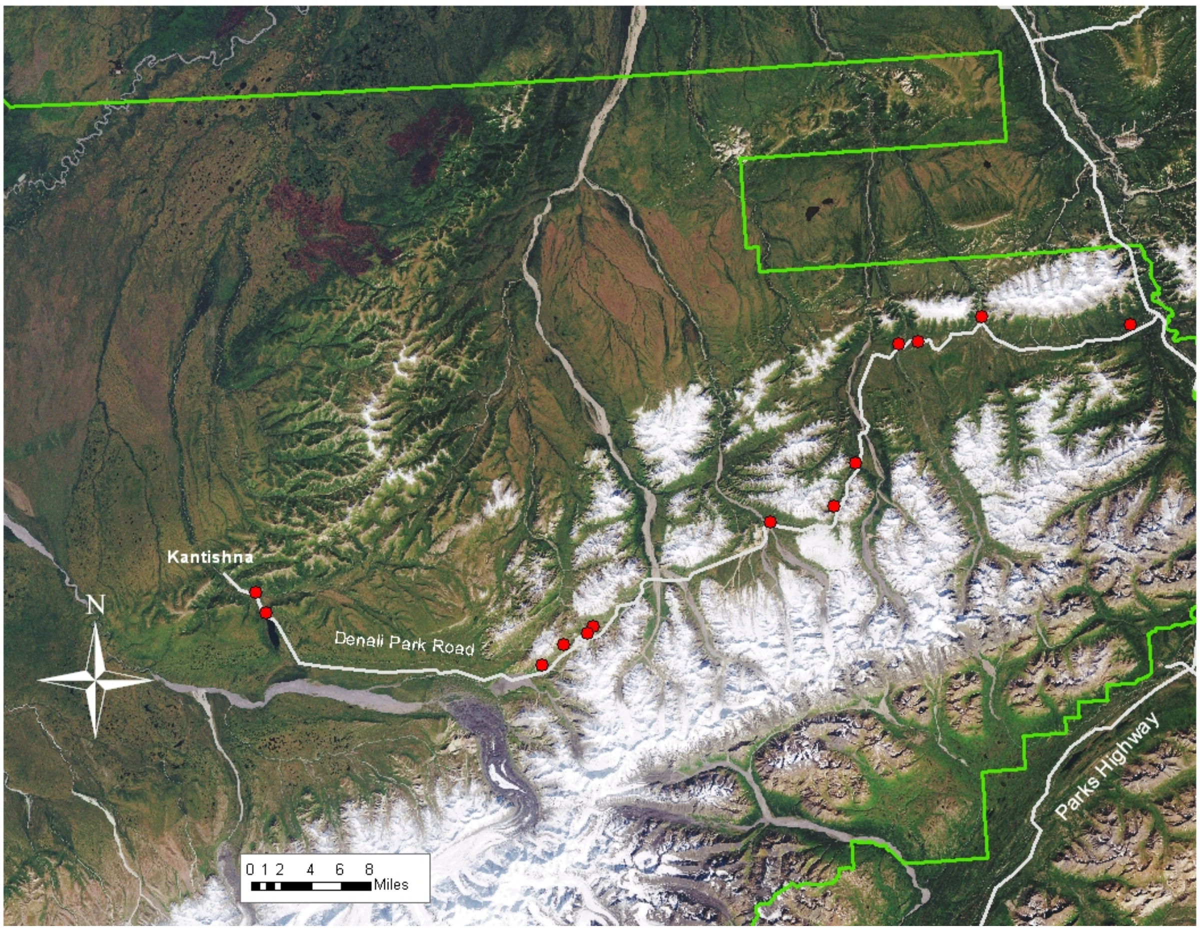

Figure 1. Locations of 2009 study sites in DENA. The easternmost site is Rock Creek; the westernmost is Moose Creek at the Kantishna bridge.

GRTS site selection

The National Hydrography Dataset (NHD, see Simley and Carswell Jr. 2009) was used as the base data for selection of the synoptic sites using the GRTS design algorithm (Stevens and Olsen 2004). The NHD for Alaska has not been edited extensively for either topological consistency or accuracy to reality; as such it is not currently appropriate to use as a basis for site selection. Accordingly, the first step was to edit the NHD for WRST to ensure topological consistency; editing for accuracy was not possible at the park scale. A Geographic Information System (GIS) was used to edit the NHD as follows. GIS datalayers for the three 4-digit hydrographic units that encompass WRST (1901, 1902 and 1904) were first merged, and then clipped to the park boundary using ArcGIS Desktop 9.3 (ESRI, Redlands, CA). The RivEX network analysis tool (Hornby 2009) was used to evaluate the NHD for topological errors (*e.g.*, disconnected polylines). Corrections to the network were effected using tools available in ArcGIS. The RivEX tool was then used with the edited and corrected NHD layer to attribute each stream segment with Strahler stream order (Strahler 1952). Stream order can be used as a surrogate for stream size; because the program is largely limited to sampling wadeable streams, it was important to

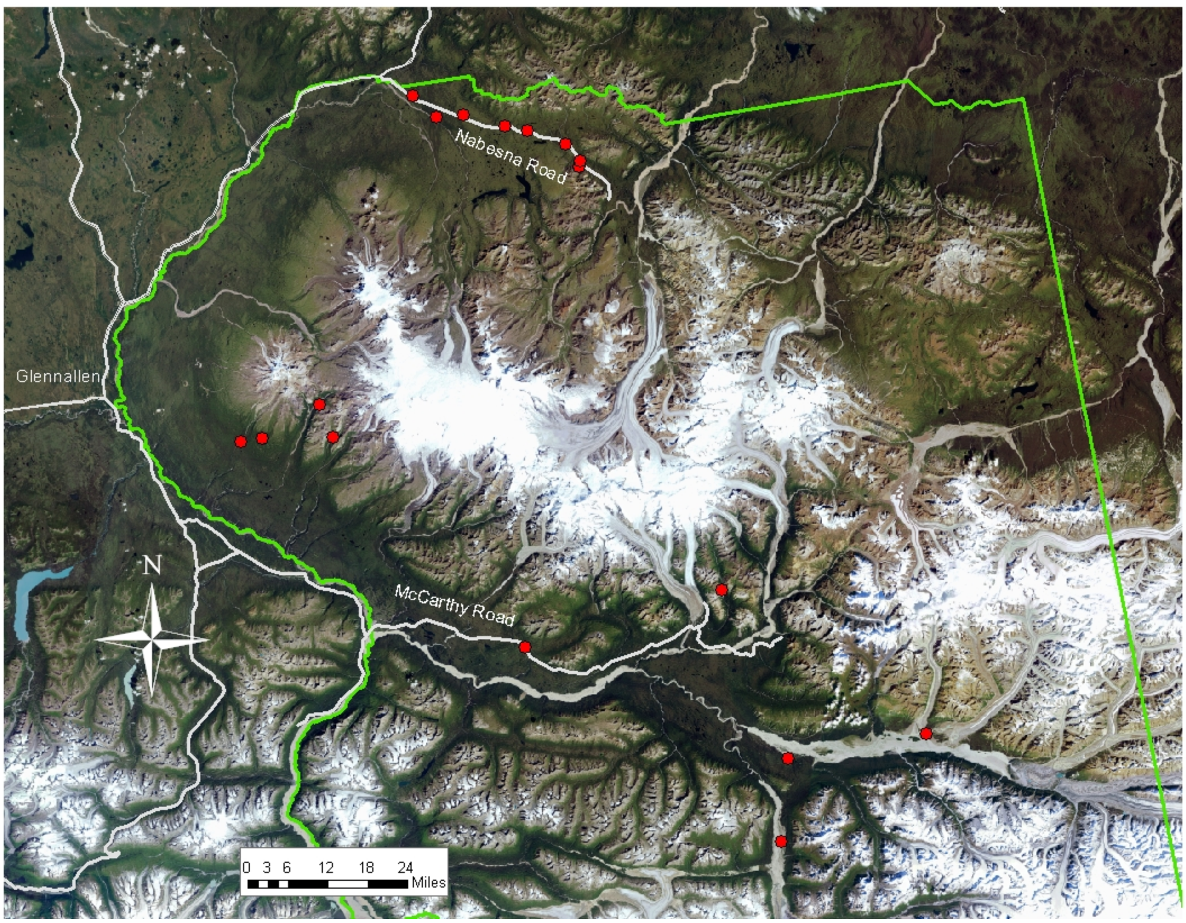

Figure 2. Location of 2009 study sites in WRST. The easternmost site is a tributary of Clear Stream, near the toe of the Hawkins Glacier; the westernmost site is the Nadina River, the southernmost site is a tributary of the Tana River.

over-represent stream segments likely to wadeable in the final site list. Although catchment area (upstream contributing area) is generally a better surrogate, it is not feasible to calculate the catchment area for every potential sampling site in even a small portion of a stream network. WRST contains over 1,000,000 acres of inholdings, mostly owned by local native corporations. Following the recommendation of WRST staff, the CAKN eliminated all stream segments that were contained within inholdings from consideration for inclusion on the list of potential sampling sites. Because the cost of accessing remote sites imposes a major limitation on the monitoring program, we attempted to stratify the sampling population by accessibility. The plan was to select 50% "accessible" sites and 50% "inaccessible" sites, as defined by the likelihood of needing a helicopter to get to the site. If trends in key indicators were coherent between the two strata, we could be confident in the future that an emphasis on accessible sites would not detract from our ability to draw conclusions about sites across the entire park or network.

Several data layers in ArcGIS were used to generate an approximate "accessibility surface". Layers delineating the locations of roads, ATV trails, major river corridors, landing strips and floatplane-accessible lakes were merged to create a layer representing potential points of access. The access point layer was then buffered at 1.0 miles to create a layer representing "accessible" sites. In retrospect, the inclusion of major river corridors was a miscalculation. The idea was that these corridors would provide access either by boat or by fixed-wing aircraft (due to the likely presence of large gravel bars that might serve as landing strips); however, in most cases it turned out that use of a helicopter would be required to access these sites.

Once the WRST NHD layer had been attributed with Strahler stream order and accessibility, we used the spsurvey package (Kincaid 2008) in the R statistical software environment (R version 2.8.1, the R Foundation for Statistical Computing, 2008) to generate the GRTS sample. The algorithm was set to select a 50-site accessible stratum with a 150-site oversample, and a 50-site inaccessible stratum with a 150-site oversample. The large oversample was chosen because it was deemed likely that the majority of sites would not be sampleable, making it difficult to identify a set of 100 sampleable sites without a substantially longer starting list of possibilities. The unequal weighting function was used to over-represent 2nd and 3rd order stream segments in the site list. All 400 sites were then examined using a variety of remotely sensed and GIS data to determine whether they were part of the target population.

Reach definition

Sampling reaches were defined using guidelines from the EPA's EMAP Wadeable Streams Assessment (USEPA 2004), and modified as necessary. A sampling reach was defined as 40 times the mean wetted width of the stream, based on five equally spaced measurements at the bottom of the proposed reach. Although this length was initially chosen as the minimum sufficient to adequately capture fish community composition in wadeable streams (Reynolds *et al.* 2003), it is also generally long enough to include a complete meander bend, which is a fundamental unit of stream geomorphology. Hence, a reach sufficiently long to encompass a meander bend should adequately capture the habitat complexity of that section of stream (Kaufmann *et al*. 1999). The minimum sampling reach length was set at 150 meters, and the maximum at 500 meters (the latter for feasibility and safety reasons). Reaches were selected to be as representative as possible of the stream section in which they were embedded; in addition, major tributary junctions were avoided and reaches near road crossings were located so as to begin at least 50 meters upstream. Once defined, the reach was subdivided into ten equally spaced sections by the placement of 11 cross-sectional transects (A – K). These transects formed the framework around which the bulk of biological and physical sampling occurred. Starting in 2007, the program dispensed with the requirement to measure and flag the transects. This procedure can take upwards of 30 minutes, which is a substantial portion of the time spent at each site. Instead, I elected to estimate the inter-transect distance by stepping it off in approximately 1-m intervals. Although this results in some variability in transect location between different sampling efforts (*e.g*., macroinvertebrates *vs*. channel geometry), the loss of accuracy is minimal because the goal of the program is to characterize the reach, rather than to establish monumented cross sections; therefore, the exact locations at which data are collected should not matter.

Table 1. List of sites sampled in 2009 with location and brief description. A * indicates sites that have been sampled at least once in previous years.

Site	Park	Description
Rock Creek*	DENA	high-gradient, confined-channel step-pool stream (LTEM site)
Travertine stream*	DENA	small stream with extensive mineral deposits near Stony Creek
Sanctuary River*	DENA	large glacially-influenced river (LTEM site)
Moose Creek at bridge*	DENA	large plane bed stream in Kantishna Hills (LTEM site)
Little Stony Creek*	DENA	high-elevation tundra stream (LTEM site)
Gorge Creek spring*	DENA	small spring-fed tundra stream below Gorge Creek overlook
Igloo Creek*	DENA	large forested stream (LTEM site)
Tattler Creek*	DENA	small steep braided stream (LTEM site)
E.F. Toklat River*	DENA	large turbid braided glacial river (LTEM site)
Hogan Creek*	DENA	small groundwater-fed stream (LTEM site)
Savage River*	DENA	large river with some glacial influence (LTEM site)
E.F. Toklat tributary*	DENA	clearwater tributary to E.F. Toklat River (LTEM site)
Lake Creek*	DENA	Wonder Lake outlet stream
Stony Creek	DENA	steep step-pool unconfined channel stream
Chalk Creek*	WRST	clearwater lake outlet stream along Nabesna Road
Rock Creek*	WRST	clearwater stream along Nabesna Road
Rock Creek tributary*	WRST	very small DOC-rich forest stream
Jack Creek at bridge*	WRST	large lake outlet stream along Nabesna Road
Little Jack Creek*	WRST	clearwater stream along Nabesna Road
Gilahina River*	WRST	large clearwater stream along the McCarthy Road
Rufus Creek*	WRST	stable groundwater stream along Nabesna Road
Nadina River	WRST	large steep turbid glacial stream near Mt. Drum
McCarthy Creek	WRST	large turbid glacial stream
Jack Creek tributary	WRST	very steep unconsolidated intermittent stream
Caribou Creek	WRST	intermittent stream along Nabesna Road
Tanada Creek	WRST	Tanada Lake outlet, large forest pool-riffle stream
Tana River tributary	WRST	large mostly sand-bottom floodplain stream
Nadina River tributary	WRST	high-elevation tundra pool-riffle stream
Sheep Gulch	WRST	high-elevation step-pool unconfined stream
Clear Stream tributary	WRST	mostly sand/gravel bottom glacial floodplain stream (clear)
Dadina River tributary	WRST	mostly sand/gravel bottom intermittent stream
Hunter Creek	WRST	large mostly sand/gravel bottom floodplain stream

Biological sampling

Biological sampling protocols were largely adopted from the Environmental Monitoring and Assessment Program's (EMAP) Wadeable Streams Assessment Field Protocols (USEPA 2004) and from methods developed at the Western Center for Monitoring and Assessment of

Freshwater Ecosystems at Utah State University (Hawkins *et al*. 2003). Macroinvertebrate samples were collected as follows: a modified net that combines elements of Surber and D-net samplers with a 500 μm mesh was used. At each transect, the net was placed in the left 1/3, center 1/3 or right 1/3 of the stream width (within these broadly defined areas, the exact sampling locations were haphazard), 1 m upstream of the first transect. The position of the first placement was determined by rolling a die, and net placements at subsequent transects followed the pattern left-center-right-left…*etc*. An area of 0.09 m^2 in front of the net opening

Figure 3. Collecting macroinvertebrates using a modified D-net (Rufus Creek, WRST).

(as defined by a hinged frame that could be lowered to the stream bed) was thoroughly searched for macroinvertebrates by individually rubbing cobbles in front of the net opening and subsequently disturbing the remaining substrate by raking to a depth of approximately 10 cm. A total of eight macroinvertebrate samples were collected and composited into a single reachwide sample. This sample represents a total of 0.72 m^2 of streambed. At some sites, this methodology was modified as follows. For "occupancy"-type samples (quasi-replicated samples that are designed to be used in the development of occupancy models), each adjacent pair of 0.09 m^2 subsamples was composited, for a total of four quasi-replicated samples, each representing 0.18 m^2 of streambed. Macroinvertebrates and organic detritus were separated from cobble and gravel and preserved in 70% ethanol. Macroinvertebrates were sorted and identified to the lowest practical taxonomic level, generally genus, by Mike Cole, a taxonomist for ABR, Inc. Due to variability within and among samples in the taxonomic resolution that could be achieved, the CAKN has established a set of Operational Taxonomic Units (OTU) to assure that taxa are not double (or triple) counted (Simmons, unpublished). In some cases this required deleting coarse-resolution data (generally family level); in other cases, higher-resolution data (species or genus) were collapsed back to coarser resolution (genus, sub-family or family). The net result, often referred to as OTU richness, is therefore a conservative estimate of the true richness at a given site.

Benthic diatoms were collected as follows. At each of the eight transects where macroinvertebrates were collected, an appropriate cobble was haphazardly selected along the same cross section used for macroinvertebrate sampling (1 m upstream of the transect itself). Cobble selection was shifted one "unit" to the right (*i.e.*, if macroinvertebrates were collected in the center of a given transect, a cobble was selected on the right). A defined area (12 cm^2) of each cobble was scrubbed and scraped to remove diatoms and the material collected was composited into a single reachwide sample. The total volume (diatoms plus rinse water) was recorded and a 40 mL subsample was removed and preserved with 2 mL Lugol's solution. At some sites, this methodology was modified as follows to allow for quasi-replicated "occupancy"-type samples to be collected. At these sites, diatoms from each adjacent pair of cobbles were composited, for a total of four quasi-replicated samples, each representing 24 cm^2 of scraped cobble. The total volume of each quasi-replicate was recorded, and a 10-mL subsample was removed and preserved with Lugol's solution. Benthic diatoms were identified to the lowest practical taxonomic level, generally species, by Julia Eichmann, a diatom taxonomist for Ecoanalysts, Inc.

Fish were captured by a combination of electrofishing, minnow trapping and angling (fly fishing). We used a Smith-Root Model LR-24 backpack electrofisher. When electrofishing, crews worked upstream through the reach. All likely habitats were fished, and we generally had two netters attempting to collect stunned fish. When minnow trapping, we deployed 6 traps in likely habitats (log jams, backwaters, undercut banks) immediately upon arriving at the sampling reach. Traps were baited with a small amount of treated salmon eggs in small perforated plastic bags, which were attached with twisty-ties to the midpoint of the trap. Traps were left in place for a minimum of 2 hours; whenever possible, they were deployed for longer periods, up to 16 hours (overnight). Captured fish were anesthetized with a mixture of clove oil and ethanol, identified and fork lengths were recorded. After completing identification and measurements, fish were allowed to recover in a bucket of fresh water before being released. Sampling by angling consisted of fly fishing the reach using a combination of wet and dry flies. The time spent angling was generally short (less than 30 minutes). Fish caught by angling were identified to species, their length was estimated and they were released immediately.

Physical and chemical data collection

We used a YSI QS650 sonde to collect temperature, specific conductivity, pH and dissolved oxygen *in situ*. Data were collected in riffles or runs and generally in midstream at the bottom of the reach. In addition, water chemistry samples were collected for later laboratory analysis. The samples were collected at the same location as the *in situ* water chemistry. A 1-l bottle was rinsed three times with stream water, and then used to collect the main sample. A 500-mL unfiltered sample for total nitrogen and total phosphorous was transferred into an acid-washed Nalgene sample bottle and placed in a cooler with dry ice to freeze. The other 500 mL was filtered through a 0.45 μm filter (previously rinsed with 50 mL of stream water). A 250-mL aliquot (for nitrate, ammonium, phosphate and common ions) was transferred to an acid-washed Nalgene bottle and placed in a cooler with dry ice to freeze. A 125-mL aliquot of filtrate (for dissolved organic carbon and silicon analysis) was transferred to an acid-washed Nalgene bottle and kept on ice. In some cases it was not possible to freeze the samples in the field; in these cases, the samples were kept cool in stream water and frozen as rapidly as possible (generally within an hour or two). Analytical water chemistry data were provided by the Cooperative Chemical Analytical Laboratory, established by memorandum of understanding no. PNW-82-187 between the U.S. Forest Service Pacific Northwest Research Station and the Department of Forest Ecosystems and Society, Oregon State University. This laboratory specializes in low-level detection of ambient stream water solutes.

Figure 4. Filtering water chemistry samples in the field (Independence Creek, WRST).

Physical data collection protocols were largely based on EMAP WSA protocols (USEPA 2004). At each transect, we measured depth (five measurements), width (wetted and bankfull), channel height (bankfull and incised), undercut banks, canopy cover (six measurements) and substrate size class (using a gravelometer – five measurements at depth locations). In intertransect segments, we measured thalweg depth and habitat type (ten measurements), width (one measurement), substrate (five cobbles along width measurement cross-section), woody debris (by size class) and fish cover (macrophytes, filamentous algae, boulders and undercut banks (qualitative estimate of extent). We measured reach slope using a transit level at some sites. We also measured discharge at a subset of sites using a Marsh-McBirney flowmeter and a topsetting wading rod.

Continuous data collection

Temperature-recording dataloggers were installed in 13 streams in May and early June (Table 2). In addition, two continuously recording pressure transducer/temperature recorders were installed at the Jack Creek site (WRST). Dataloggers were generally secured using vinyl-coated steel cable to attach them to riparian trees. Loggers were additionally wired to the underside of large anchor rocks to keep them in place. Where possible, loggers were placed in relatively deep, fast–flowing water (*e.g.*, in a run). Care was taken to minimize the visibility of both the logger and the anchoring system. Two other temperature loggers, both installed at remote sites in 2008, could not be retrieved before the end of the 2009 field season, and remain in place, or so it is hoped. Loggers were set to record temperature every 30 minutes. This resulted in approximately an 18 month maximum lifespan for the data record. An 18-month record is adequate when loggers are installed and retrieved every year or so; however, in cases where loggers are left over multiple winters, data may be lost. In future years, loggers will be set to record temperature once per hour – this should provide enough detail regarding diurnal variation while allowing over 3 years of data to be collected.

Table 2. List of sites where temperature loggers were installed in 2009, along with brief site descriptions.

Site	Park	Description
Sanctuary River	DENA	large glacially-influenced river (LTEM site)
Moose Creek at bridge	DENA	large clearwater river in Kantishna Hills (LTEM site)
Little Stony Creek	DENA	high-elevation tundra stream (LTEM site)
Lake Creek	DENA	Wonder Lake outlet stream
Tattler Creek	DENA	small steep braided stream (LTEM site)
Igloo Creek	DENA	large forested stream (LTEM site)
Hogan Creek	DENA	small groundwater-fed stream (LTEM site)
E.F. Toklat tributary	DENA	clearwater tributary to EF Toklat River (LTEM site)
Chalk Creek	WRST	clearwater stream along Nabesna Road
Rock Creek	WRST	clearwater stream along Nabesna Road
Jack Creek	WRST	large lake outlet stream along Nabesna Road
Caribou Creek	WRST	intermittent stream along Nabesna Road
Rufus Creek	WRST	stable groundwater forest stream
Solo Creek (2008)	WRST	clearwater tributary to White River
Tana River tributary (2008)	WRST	Deep, very low-gradient stream in Tana River floodplain

Data Analysis

Initial data manipulations, QA/QC and descriptive statistics were conducted using MS Excel 2007. Power analysis was conducted using the G*Power software package (Erdfelder *et al*. 1996). Exploratory multivariate analyses were conducted using PC-ORD 6.0, a multivariate statistics package designed for community ecology (McCune and Medford 2011). Cluster analyses for RIVPACS models were also conducted in PC-ORD 6.0. Other analyses for RIVPACS modeling, including random forest discriminant analysis, were conducted using R scripts developed by researchers at the E.P.A. and other institutions (J. Ostermiller, personal communication). Individual analytical methods are discussed in the Results and Discussion section as appropriate.

Results and Discussion

Water Chemistry

The water chemistry results (Table 2) revealed a very wide range of conditions across the two parks, consistent with results from previous years. Descriptive statistics derived from all of the water chemistry data collected from 2006-2009 are generally similar to those calculated in 2009 alone. In general the maximum and minimum values for a given constituent vary from year to year much more than the mean and median values, as would be expected. With four years of water chemistry data in hand, it is now possible to start looking at spatial and temporal patterns in some of the constituents. Many chemical constituents vary over several orders of magnitude across the Central Alaska Network. For example, specific conductivity varies from a low of 27 μS/cm (tributary of the Nadina River in WRST, 2009) to a high of 1268 μS/cm (Rock Creek in DENA, 2009), a nearly 50-fold range. Sulfate is even more variable, ranging from 0.14 mg/L (tributary of Rock Creek in WRST, 2007) to over 147 mg/L (Rock Creek in WRST, 2009), greater than a 1000-fold range. Even more remarkable is that these two sampling sites are less than 10 meters apart.

Table 3. Summary water chemistry statistics from 2009 samples.

	Minimum	Mean	Median	Maximum
Temperature	1.1°C	6.1°C	5.0°C	16.6°C
Specific Conductance	27 μS/cm	400 μS/cm	418 μS/cm	1268 μS/cm
pH	6.5	7.96	8.01	8.47
Alkalinity (HCO_3-C)	3.3 mg/L	24.4 mg/L	22.5 mg/L	77 mg/L
Total nitrogen	3 μg/L	164 μg/L	155 μg/L	390 μg/L
Nitrate-N	<1 μg/L	99 μg/L	106 μg/L	280 μg/L
Total phosphorous	3 μg/L	20 μg/L	13 μg/L	128 μg/L
Phosphate-P	< 1 μg/L	2 μg/L *	<1 μg/L *	22 μg/L
Sulfate-S	0.14 mg/L	30 mg/L	20 mg/L	147 mg/L
Dissolved organic carbon	0.33 mg/L	1.5 mg/L	1.0 mg/L	14 mg/L

*90% of samples tested in 2009 had phosphate-P levels at or below the level of detectability reported by CCAL. I substituted values of ½ the detection limit of 1 μg/L for calculating descriptive statistics, although with the understanding that it may introduce error (Helsel 2006).

Another feature of these streams is the generally low levels of phosphate-P. The mean concentration of phosphate-P among all samples collected from 2006-2009 was less than 2 μg/L, and 60% of samples had undetectable levels (<1 μg/L). Total phosphorus (TP) was also generally low, with a few exceptions. The mean TP concentration in these streams (20 μg/L) is more than four-fold lower than the comparable value (90 μg/L) for wadeable streams in the lower 48 (USEPA 2006). The median concentration for CAKN streams is only 13 μg/L, which is strikingly low. In fact, only a handful of streams in the CAKN had a TP concentration higher than 50 μg/L – most of these are glacial streams with high sediment loads and very low levels of dissolved phosphate. This suggests that suspended sediment was the source of the high TP levels and that much of it is not biologically available. The generally low phosphate levels in CAKN streams are reflected in N:P ratios, which give an indication of nutrient limitations in aquatic ecosystems. In general, N:P ratios greater than 16 (the "Redfield Ratio") are taken to indicate phosphorous limitation of aquatic primary production, whereas ratios less than 16 suggest nitrogen limitation (*e.g.*, Dodds 2002). According to this rule of thumb, most streams in the CAKN are severely P-limited, as the median N:P ratio observed in samples collected thus far is 211:1, and the range goes as high as 2000:1 (Swift Creek in WRST). Only 14 (of 123 total) samples with N:P ratios less than 16 were collected. No obvious patterns emerge that explain the distribution of N:P ratios. For example, the lowest (Nadina River, 0.55) and one of the highest (E.F. Toklat River, 998) ratios were observed in glacial rivers. Within-site temporal variation in N:P ratio was also very high. An important caveat, however, is that much of this variation is driven by differences in phosphate-P. Because for most samples, phosphate-P was near the limit of detection, the variation in N:P, both within and among sites, is probably exaggerated somewhat.

Table 4. Temporal variation in chemical constituents from CAKN streams. Statistics were calculated using data from streams with at least 3 (Alkalinity, total nitrogen, nitrate-N, total phosphorous, sulfate-S, dissolved organic carbon) or 4 (Specific conductance, pH) measurements from different years. Variation is expressed as the coefficient of variation (CV). Minimum and maximum values represent variation at individual streams, whereas the mean value is the average CV across all streams.

	Minimum CV	Maximum CV	Mean CV
Specific Conductance	0.02	0.43	0.23
pH	0.014	0.05	0.03
Alkalinity (HCO_3-C)	0.13	0.77	0.39
Total nitrogen	0.028	1.4	0.43
Nitrate-N	0.014	1.1	0.49
Total phosphorous	0.18	2.5	0.78
Sulfate-S	0.27	1.3	0.63
Dissolved organic carbon	0.05	1.4	0.54

An ongoing challenge for the program will be dealing with the extremely low concentrations of many important solutes in these streams. This is despite the very low detection limits of the CCAL methods for most solutes. The most problematic of these are soluble phosphate-P (aka soluble reactive phosphorous - SRP) and ammonium-nitrogen. Of the 123 samples so far analyzed, 60% are below the level of detection for SRP (1μg/L) and a full 89% are below the level of detection for ammonium nitrogen (10 μg/L). Statistical methods based on survival analysis exist for dealing with these so-called “non-detects”, generally by selecting values from a modeled distribution using maximum likelihood estimation techniques; however, they do not generally perform well for small data sets (less than 30-50 detected values), where there is insufficient evidence to determine whether the assumed distribution fits the data adequately (Helsel 2006). In these cases, we may have to use non-parametric procedures designed for censored data, the software algorithms for which are currently not adapted to deal with environmental data (Helsel 2005, 2006).

A final note concerns total dissolved organic nitrogen (DON). DON comprises a substantial portion of the total nitrogen transported by pristine streams like those that characterize the CAKN, and a large proportion of it can be bioavailable (*e.g.*, Kaushal and Lewis 2005, Scott *et al*. 2007). Hence, understanding variation in DON both within and among streams in the CAKN is important to a more comprehensive understanding of both landscape patterns and long-term change in ecosystem structure and function. The most straightforward way to quantify DON is indirect. Total dissolved nitrogen (TDN) is relatively simple to assess, and is composed of DON, nitrate-N and ammonium-N. Thus, DON can be quantified by subtracting [nitrate-N + ammonium-N] from the TDN fraction. For this reason, the CAKN began requesting TDN analysis in 2009. However, the TDN results were problematic in that for most samples, TDN was higher than total nitrogen (TN), suggesting that the filtered samples were contaminated. Further investigation revealed that the probable source of the contamination was the nitrocellulose filters themselves. In future years, polycarbonate or other nitrogen-free media will be used to filter the samples for TDN determination.

Macroinvertebrates

A total of 31 macroinvertebrate samples were collected from nine DENA streams (14 site visits) and 13 WRST streams (17 site visits) in 2008. As was the case in previous years, observed OTU taxa richness varied tremendously among the sampled sites, ranging from four unique taxa at the Nadina River in WRST to 37 at a tributary of the Tana River in WRST. The mean richness across all samples collected in 2009 was 22.4 taxa, which is somewhat higher than was observed in previous years. Chironomid midges accounted for approximately 34% of taxa richness across all sites, and 39% of individuals. This is substantially lower than reported by Oswood (1989), who found that chironomids constituted on average 59% of individuals in Alaskan streams, but is similar to the results observed in CAKN streams since 2006. Densities varied from a low of 19 individuals/m^2 at the East Fork Toklat River in DENA (August sample) to a high of over 47,000 per m^2 at Rufus Creek in WRST, with a mean of 4255 individuals/m^2.

Table 5. List of aquatic insect taxa identified from 2009 CAKN samples and not reported in Milner *et al*. (2003) or Conn (1998). Milner *et al*. sampled 58 streams parkwide.

Plecoptera (stoneflies)	
	Diura sp.
Diptera (true flies)	
	Unidentified Empididae
	Stratiomyidae (family)
	Phoridae (family)
	Limnophila sp.
	Pseudokiefferiella sp.
	Synorthocladius sp.

In 2007 and 2008 a total of 31 aquatic invertebrate taxa not reported by Milner *et al*. (2003) or by Conn (1998) were collected by the CAKN crew in DENA (see Table 5 in Simmons 2010). In 2009 we collected an additional seven previously unreported insect taxa (Table 5), including two new families (Stratiomyidae, also known as soldier flies, and Phoridae). This brings the 3-year total to a minimum of 38 newly reported aquatic macroinvertebrate taxa, which is remarkable given the limited geographic and temporal scope of the CAKN program in DENA and reinforces the argument, made following the 2008 field season, that the streams and rivers of DENA remain woefully understudied,. As was discussed last year (Simmons 2010), the true total is likely to be substantially higher, as we were able to identify to the species or genus level a number of taxa reported by Milner *et al.* as identified only to the genus or family level, respectively. A similar situation no doubt entails in WRST, but there is no comparable literature available for comparison in that park unit.

A total of 143 unique macroinvertebrate taxa have now been identified in the three years of the CAKN program. It is likely that we will continue to add to this total for a number of years given the relative paucity of stream macroinvertebrate data from these parks, particularly from WRST. Three of these taxa have never been formally identified, and would represent new species, or potentially new genera. One of these unidentified taxa has been collected at multiple sites since 2007. This previously unidentified midge of the subfamily Orthocladiinae, first identified in samples from 3 sites in 2007, and collected again at 3 sites in 2008, was also collected at 3 sites in 2009. The midge has been collected in all 3 years from Hogan Creek (DENA), and in both 2008 and 2009 in the Travertine Stream (DENA). In 2009 it was also collected in Little Stony Creek (DENA), in both the summer and fall samples, but was not found in any samples from WRST. The previously unidentified midge from the subfamily Tanytarsini that was collected from a stream in WRST in 2008 was not found in any 2009 samples. Lastly, a previously unidentified member of the family Empididae (dance flies) was collected at 7 sites in 2009. It is likely that this taxon was collected in previous years as well, but was reported as *Neoplasta*, to which is superficially similar. The specimens collected in 2009 were submitted to John MacDonald at Purdue, one of the world’s leading empidid specialists, but he was unable to determine its identity. Although it would be satisfying to identify and name these new species, the necessity of capturing live larvae and rearing them to the adult stage in the laboratory for identification means this next step is unlikely to occur soon.

In 2007, the density of macroinvertebrates in Igloo Creek below the Denali Park Road construction - 275/m^2 - was only 3% of that observed at the Igloo Creek site above the construction - 8,000/m^2 (Simmons 2009). In 2008, with road construction complete, the lower site appeared to have recovered somewhat, with a macroinvertebrate density that was 10 times higher than in 2007 (2807 per m^2) and nearly 28% of the observed density at the upper site. Densities at the upper site were comparable in both years (8,000 vs. 10,000 per m^2), suggesting that the road construction may have had a negative, albeit apparently temporary, effect on macroinvertebrates downstream. In 2009, macroinvertebrate density at the lower site was slightly higher than in 2008 (3234/m^2), suggesting that recovery is continuing.

Benthic Diatoms

30 diatom samples were collected from 25 unique sites in 2009. As was observed in previous years, species richness for diatoms was substantially higher than for macroinvertebrates, and varied from a low of one species (Nadina River in WRST) to a high of 77 species (Jack Creek in WRST), with a mean of 44 species per site. A total of 396 diatom species have now been identified through the CAKN program. One hundred sixty-six of these were first identified in samples collected in 2006, 68 additional species were first identified in the 2007 samples, 126 in 2008, and another 36 in 2009. In general, the lowest species richness and density were found in unstable braided or glacial systems, and the highest in stable spring-fed creeks and lake outlets. Although we continue to find that species richness and densities are lowest in turbid and unstable streams, species richness in some of these systems has been remarkably high. For example, the East Fork of the Toklat River has consistently supported 12 species of diatoms during the summer, as opposed to only three to five macroinvertebrate taxa. Even more surprising was the collection in 2009 of 49 diatom species from McCarthy Creek, a turbid glacial river, albeit one

with a single channel and relatively stable banks. More consistent with expectations was the Nadina River in WRST, a high-gradient unstable glacial river where only a single diatom species was collected, and that at very low density.

Didymosphenia geminata, an emerging invasive species with a number of unusual properties, is native to boreal streams, but has been observed recently to cause problematic blooms even within its native range. *D. geminata* was identified at eight of the 25 sites sampled (one remote sites in WRST and seven sites along the park road in DENA), albeit at generally low densities. These findings are consistent with those from previous years. A nuisance bloom was observed in the East Fork Toklat tributary in DENA in 2009; a sample of the bloom material was collected and submitted to Dan Bogan and Dan Rinella at the University of Alaska Anchorage, who confirmed its identity as *D. geminata*. Interestingly, this site and the site where a nuisance bloom was observed in 2008 (Igloo Creek) are in adjacent basins. The CAKN program will continue to monitor the distribution and abundance of this species, and we are developing a set of procedures to allow other field personnel in the parks to recognize and collect samples of suspected *D. geminata* blooms.

Glacial river phenology

2009 marked the second year in which CAKN sampled a glacial river in both the summer and the fall. Glacially-influenced rivers are typically turbid and highly dynamic during the summer melt season, but clear up and stabilize in the fall. This clearwater phase, which occurs in both fall and spring, may provide a critical subsidy to stream consumers, including fish, as the stable hydrology and clear conditions should allow for a burst of primary productivity during the time period after the end of the glacial melt season, but before the angle of the sun, temperature and day length decrease too much (and *vice versa* in the spring). Because climate warming may alter these processes, the CAKN monitoring program is determined to include multi-season monitoring of glacial rivers as an important component. The East Fork of the Toklat River in DENA was chosen as the first glacial river to be monitored in this way. In 2007, 2008 and 2009, the river was sampled in late August, when it was turbid, and exhibited very dynamic hydrology, and then in 2008 and 2009 sampled again in late September, when the water was clear and the flow was low and stable.

The results have been remarkable, and are shown in Table 5. It is not known exactly when the melt season ended for the East Fork Glacier in either year, but given observations of other glaciers in the network, it was likely not more than two weeks before the fall sampling in September. Hence, most if not all of the changes discussed here as differences between “summer” and “fall” likely occurred within this short time period. With the exception of total phosphorous, there is little change in water chemistry between summer and fall. The high level of total phosphorous observed in the summer is almost certainly related to phosphorous in the sediment load carried during glacial melt. High total phosphorous accompanied by low soluble reactive phosphate is typical of the turbid glacial rivers that have been examined in the Central Alaska Network. The likely explanation for this lack of interseasonal variability is a significant contribution to the shoulder season baseflow from groundwater sources with a substantial glacial component. Despite the minimal effect on water chemistry, however, a significant biological change occurs over this short time period. Although diatom species richness appears to decrease by an average of about 30%, diatom density increases by over an order of magnitude (Table 6). The change in species richness is interesting, as opposite patterns were observed in 2008 and

Table 6. Summary of instream conditions at East Fork Toklat River in summer *vs*. fall. Summer conditions represent a 3-year average (2007-2009), whereas fall conditions are a two-year average (2008-2009).

Metric	Summer value	Fall value
Temperature	7.3°C	2.7°C
Specific conductivity	426 μS/cm	460μS/cm
Total nitrogen	200 μg/L	190 μg/L
Total phosphorous	114 μg/L	15 μg/L
pH	8.3	8.2
Nitrate-N	147 μg/L	155 μg/L
Soluble reactive phosphorous	<1 μg/L	<1 μg/L
Macroinvertebrate taxon richness	4	7.5
Macroinvertebrate density	11 / m^2	44 / m^2
Diatom species richness	21	13.5
Diatom density	37,500/cm^2	493,244/cm^2

2009. In 2008, species richness was higher in the fall (15) than in the summer (12), whereas in 2009 richness was higher in the summer (30) than in the fall (12). The former pattern is what one might expect *a priori*; it will be interesting to see whether either pattern becomes typical over the long run. The observed change in the macroinvertebrate community is even more remarkable, given the typically much longer life cycles and slower growth rates of macroinvertebrates relative to diatoms. Macroinvertebrate taxon richness nearly doubles, and macroinvertebrate density increases by 400%, although it remains low relative to nonglacial streams. These results support the idea that the shoulder seasons are important to overall stream productivity in glacial systems, and as the timing and duration of these clearwater periods are likely to be altered by ongoing climate change, the CAKN program plans to expand its two-season monitoring of glacial streams and rivers.

Re-evaluating the comparison between LTEM and CAKN macroinvertebrate data

An important focus for the 2007 field season was to test whether legacy invertebrate data collected in Denali National Park and Preserve (DENA) as part of the Long Term Ecological Monitoring Program (LTEM) would be compatible with data collected using the Central Alaska Network (CAKN) stream monitoring protocols. Under the direction of Sandy Milner, invertebrate samples have been collected annually since 1994 at 14 sites along the Park Road. Furthermore, the program collected macroinvertebrate data in 1994-1996 from a total of 58 streams located throughout DENA. These data should provide an invaluable starting point for the determination of long-term trends in the ecology of DENA streams; however, differences in the field and laboratory methods used by Milner and the CAKN program have the potential to complicate efforts to synthesize these two data streams. Initial comparative analyses based on data collected by both groups at 10 sites along the park road in 2007 revealed substantial differences in the results produced by the two programs, in both reported taxa richness and community composition, at these 10 sites. In 2008, CAKN resampled eight of those 10 sites to assist in the re-evaluation of these findings and in 2009 a similar but overlapping set of seven of these sites was sampled again (Table 7). The 2007 findings can be summarized as follows. There were two cases of obvious taxonomic disagreement; in both cases the CAKN taxonomist made a strong case that the CAKN designation was correct, and for the purposes of analysis, the assumption was made that the CAKN designation should apply to both data sets. A number of other taxonomic adjustments were also necessary to make the two datasets comparable; these were mostly the result of the higher level of taxonomic resolution in the CAKN data. After

Table 7. LTEM sites that were resampled by CAKN in 2008 and/or 2009.

Stream name (reach length in meters)	Description of CAKN sampling reach
Hogan Creek (150 m)	Reach starts ~50 m above road
Sanctuary River (500 m)	Reach starts opposite pullout ~1/4 mile past bridge
Igloo Creek (250 m)	Reach starts ~100 m above bridge
Tattler Creek (150 m)	Reach starts ~50 m above road
East Fork Toklat River *	Reach starts ~50 m above tributary confluence
East Fork Toklat tributary (150 m)	Reach starts ~200 m above confluence
Little Stony Creek (150 m)	Reach starts ~50 m above road
Moose Creek (500 m)	Top of reach at Kantishna bridge
Savage River (500 m)	Top of reach ~200 m downstream of bridge

*The E.F. Toklat is a dynamic braided river. For this type of system, we use a “transverse” reach across the floodplain and sample all braids.

making these adjustments, it was found that both mean taxa richness and mean density were substantially higher in the CAKN samples than in the LTEM samples (mean taxa richness was 35% higher, mean density was 31% higher). The differences in reported richness were highest at large, wide rivers. The most parsimonious explanation was that differences in sampling methodology largely account for this discrepancy. The LTEM samples are collected from within a 10-meter reach regardless of stream size, whereas the CAKN samples are collected from a reach that is much longer (40 times the mean wetted width, with a minimum reach length of 150 meters and a maximum of 500 meters), and therefore may capture more of the diversity of the larger stream segment. This effect should be more pronounced in larger streams, as was observed. The 2008 and 2009 CAKN data continue to support this idea in general. The mean (2007-2009) observed richness among all of the CAKN samples is 53% higher than was the case for the 2007 Milner samples (Table 8), which suggests that the 2007 results were not aberrant. In most cases, the difference is still greater for larger streams – the exception is Tattler Creek, where the mean CAKN richness is 80% higher, although Tattler Creek is one of the smaller streams in the set.

We also re-examined the species composition of the various samples. In 2007, the mean Bray-Curtis dissimilarity between LTEM-CAKN stream sample pairs (excluding the depauperate E.F. Toklat River) was 0.34, compared to the global between-sample dissimilarity of 0.63 (Simmons 2009). Bray-Curtis dissimilarity is a measure of the degree to which the taxa lists of two samples overlap. It varies from zero to one, with a value of zero indicating that two samples are identical in terms of species composition, and a value of one indicating that they share no taxa whatsoever in common. Although the mean between-sample dissimilarity (within-stream) from 2007 was about half of the mean between-stream dissimilarity, it was still higher than it should be. In 2006, the CAKN program resampled a single reach (Chalk Creek in WRST) five times in one day. The mean Bray-Curtis dissimilarity among these replicate macroinvertebrate samples was 0.19, meaning that they shared 81% of their taxa lists on average. The mean dissimilarity among all WRST streams sampled in 2006 was 0.66, which is comparable to the mean for these Denali streams. A similar exercise was conducted in 2009 at one of the LTEM streams (Little Stony

Creek) and the results were comparable, with a mean between-replicate dissimilarity of 0.18. This distance (0.18 or 0.19) then represents the approximate minimum between-sample distance

Table 8. Comparison of observed taxa richness (S) at LTEM sites in DENA 2007-2009.

Stream Name	S (Milner 2007)	Mean S (CAKN 2007-2009)
Hogan Creek (150 m)	12	12.3
Sanctuary River (500 m)	12	17.5
Igloo Creek (250 m)	11	17.7
Tattler Creek (150 m)	10	18
East Fork Toklat River *	3	4
East Fork Toklat tributary (150 m)	8	17.3
Little Stony Creek (150 m)	10	12.2
Moose Creek (500 m)	18	27.3
Savage River (500 m)	13	21
Highway Pass (200 m)	5	9
Mean across all samples	10.2	15.6

that can be expected, with the understanding that differences in the degree of spatial heterogeneity among streams may increase or decrease it. The remaining variability between the two data sets could result largely from taxonomic differences we have not yet identitifed (Stribling *et al*. 2003) or may stem from the differences in field methodology or from a combination of both. Natural variability (spatial and temporal) in macroinvertebrate community composition presents a major challenge for the detection of trends in stream ecosystems. This variability can be substantial; hence, it is critical that other sources of variability, such as those due to sampling error, be minimized.

When the data from the streams CAKN resampled in 2008 and 2009 are included in the analysis, it was found that the mean Bray-Curtis distance between CAKN stream-year pairs (0.36) was less than the mean distance between LTEM and CAKN samples collected at those 8 streams only a few days apart in 2007 (0.40). Distances between the Milner samples and both the 2008 (0.49) and 2009 (0.37) CAKN samples were similar to the 2007 results. Most of this difference was again driven by the substantial dissimilarity in samples collected in larger rivers. In contrast, in 2009 the CAKN program collected two samples from Chalk Creek 17 days apart; the Bray-Curtis dissimilarity between these two samples was only 0.17, approximately the same as the dissimilarities between both sets of same-day replicates. The substantially larger dissimilarities between the LTEM and CAKN samples suggests there is a systematic problem, at least with respect to larger streams and rivers, in comparing data collected by the two programs.

Initial analyses of year-to-year variability

Water Chemistry

2009 was the fourth year of data collection (third for most sites), which provides enough data to begin to look at year-to-year variability in some of the metrics collected. For water chemistry data, there are 27 sites with data from at least two years, 14 sites with data from at least three years, and three sites with data from all four years. Several sites have been visited multiple times in one or more years as well. Although power to detect trends depends on various factors, an

Table 9. Visit-to-visit variability in water chemistry parameters at sites sampled at least once per year from 2006-2009.

Stream	Parameter	N	Mean	95% CI	CV
Rock Creek	Specific conductance	8	458 μS/cm	+/- 345	0.75
Chalk Creek	Specific conductance	12	355 μS/cm	+/- 103	0.29
Jack Creek	Specific conductance	10	238 μS/cm	+/- 131	0.55
Rock Creek	pH	7	7.8	+/- 0.22	0.04
Chalk Creek	pH	10	8.22	+/- 0.17	0.03
Jack Creek	pH	8	7.86	+/- 0.19	0.04
Rock Creek	Total alkalinity	7	38 mg/L	+/- 14.7	0.52
Chalk Creek	Total alkalinity	9	33 mg/L	+/- 5.17	0.24
Jack Creek	Total alkalinity	7	22 mg/L	+/- 8.95	0.55
Rock Creek	Total nitrogen	7	253 μg/L	+/- 67	0.36
Chalk Creek	Total nitrogen	9	299 μg/L	+/- 172	0.88
Jack Creek	Total nitrogen	7	183 μg/L	+/- 76	0.56
Rock Creek	Dissolved organic carbon	7	3.96 mg/L	+/- 2.3	0.79
Chalk Creek	Dissolved organic carbon	8	2.69 mg/L	+/- 2.2	1.18
Jack Creek	Dissolved organic carbon	7	2.10 mg/L	+/- 0.91	0.59
Rock Creek	Total phosphorous	7	24 μg/L	+/- 14	0.80
Chalk Creek	Total phosphorous	9	200 μg/L	+/- 331	2.53
Jack Creek	Total phosphorous	7	26 μg/L	+/- 13	0.65
Rock Creek	Nitrate-nitrogen	7	106 μg/L	+/- 58	0.75
Chalk Creek	Nitrate-nitrogen	8	117 μg/L	+/- 48	0.63
Jack Creek	Nitrate-nitrogen	7	78 μg/L	+/- 54	0.93
Rock Creek	Sulfate-sulfur	7	37.3 mg/L	+/- 36	1.31
Chalk Creek	Sulfate-sulfur	9	24.7 mg/L	+/- 10.9	0.63
Jack Creek	Sulfate-sulfur	7	23.7 mg/L	+/- 3.4	0.15

estimate of the year-to-year (or visit-to-visit) variability in the value of a given metric (*e.g.*, the 95% confidence interval or the coefficient of variation) provides some insight into how difficult trend detection is likely to be. Table 9 shows the mean, 95% confidence interval and coefficient of variation (CV) for some key water chemistry parameters at the three sites for which data are available from all four years of the program. All three sites were visited more than once in at least one of the four years. Chalk Creek was visited twice in 2006 and 2007, four times in 2008 and five times in 2009; Rock Creek was visited twice in 2006, once in 2007, four times in 2008 and three times in 2009; and Jack Creek was visited once in 2006, once in 2007, three times in 2008 and four times in 2009.

These streams are reasonably similar in terms of water chemistry, which is not unexpected given their spatial proximity. Chalk Creek is a tributary of Jack Creek, and the Rock Creek site is less than 10 miles west of the Chalk Creek site, albeit in a different basin. Even with just three streams, however, it is clear that there is a great deal of variation among parameters in terms of their temporal variability, and also that the variability for some parameters itself varies substantially among streams. Specific conductance and pH show relatively low variability at all three sites, whereas dissolved organic carbon and total phosphorous are quite variable at all three sites. Sulfate, on the other hand, appears to be relatively invariant at Jack Creek (CV = 0.15), but highly variable at Rock Creek (CV = 1.31).

Statistical power to detect changes similarly will vary substantially across parameters and streams. A preliminary power analysis (with $1-\beta = 0.90$, $\alpha = 0.05$) of water chemistry at Chalk Creek based on these data suggests that detecting a moderate increase (20%) in the mean measured specific conductivity would not require an unreasonable number of measurements (40, of which 12 have already been taken). On the other hand, detecting a similar change in dissolved organic carbon would require nearly 1200 measurements. If we were only able to take 40 dissolved organic carbon measurements, the power to detect this 20% change would be less than 15%. Similar problems apply to other important solutes such as phosphorous and nitrogen. Clearly, detecting moderate change in the mean concentrations of these solutes will be challenging. However, it is important to note that a substantial proportion of this variability may be linked to measurements taken during extreme events (*i.e*. spates and droughts), and that baseflow values for some constituents may be relatively stable.

This is illustrated in Table 10 for two cases. On July 11, 2007 flow in Chalk Creek was turbid and nearly bankfull as a result of an intense, but localized, rain event the previous evening. Flow in Rock Creek was well below bankfull, but still substantially higher than normal. In contrast, the weeks preceding the July 14-18, 2009 sampling were characterized by drought-like conditions, with less than 5 mm precipitation recorded at a rain gauge we installed near the Tanada Creek trailhead (~2.5 miles southeast of the Rock Creek site, 7 miles northwest of the Chalk Creek site). The 30-year average for July is over 100 mm. Flow in Rock Creek was substantially lower than baseflow, whereas flow in Chalk Creek was lower than usual, but only moderately so. To investigate the effects of these events on streamwater chemistry and also on the variability observed in various solutes, the mean values and 95% confidence intervals for several of the measurements shown in Table 9 were recalculated with the data from July 11, 2007 and July 14-18, 2009 withheld. The results of this exercise are shown in Table 10. The baseflow values are those calculated with the high flow (July 11, 2007) and low flow (July 14-18, 2009) data withheld. With a few exceptions, the measurements made during both high and low flow are well outside the 95% confidence intervals for each of these constituents at baseflow, and this is the case for both streams. Substantial changes in stream water chemistry during high flows are common, and result largely from changes in the relative importance of different hydrologic flowpaths. In the case of Chalk Creek, the concentrations of most cations and anions dropped, along with conductivity, alkalinity and nitrate, whereas the levels of dissolved organic carbon, total nitrogen and phosphorous, ammonium and soluble phosphate spiked dramatically. This is consistent with a substantial increase in overland and shallow subsurface flow contributing to overall discharge. Similar, though less dramatic changes were observed at Rock Creek.

Low flow conditions had different, though no less substantial effects on water chemistry. These effects were most obvious for Rock Creek, probably because one of the headwater sources for Chalk Creek is a lake, which would be expected to have reduced the magnitude of the flow response to decreased precipitation. Relative to baseflow values, Rock Creek exhibited substantial increases in the concentrations of most cations and anions, along with nitrate-nitrogen and total nitrogen, whereas DOC and total phosphorous dropped markedly. Specific conductivity

Table 10. Effects of high and low flow on water chemistry in 2 WRST streams. High flow samples were collected July 11, 2007, low flow samples from July 14-18, 2009. Underlined values are outside of the 95% confidence interval for baseflow.

Stream	Parameter	Baseflow mean +/- CI	High flow	Low flow
Rock Creek	Specific conductance	347 +/- 76.5 µS/cm	<u>255 µS/cm</u>	<u>1221 µS/cm</u>
Chalk Creek	Specific conductance	363 +/- 79.6 µS/cm	<u>245 µS/cm</u>	412 µS/cm
Rock Creek	pH	7.82 +/- 0.30	7.95	7.57
Chalk Creek	pH	8.27 +/- 0.19	8.1	<u>7.99</u>
Rock Creek	Total alkalinity	33.1 +/- 9.46 mg/L	<u>23.3 mg/L</u>	<u>77.6 mg/L</u>
Chalk Creek	Total alkalinity	31.8 +/- 4.97 mg/L	27.5 mg/L	<u>46.8 mg/L</u>
Rock Creek	Total nitrogen	210 +/- 54 µg/L	<u>390 µg/L</u>	<u>330 µg/L</u>
Chalk Creek	Total nitrogen	237 +/- 73 µg/L	<u>950 µg/L</u>	<u>80 µg/L</u>
Rock Creek	DOC	3.51 +/- 0.55 mg/L	<u>9.5 mg/L</u>	<u>0.66 mg/L</u>
Chalk Creek	DOC	1.68 +/- 0.94 mg/L	<u>10.2 mg/L</u>	1.28 mg/L
Rock Creek	Total phosphorous	24.8 +/- 18 µg/L	37 µg/L	7 µg/L
Chalk Creek	Total phosphorous	36 +/- 30 µg/L	<u>1546 µg/L</u>	<u>5 µg/L</u>
Rock Creek	Nitrate-nitrogen	82 +/- 13 µg/L	<u>49 µg/L</u>	<u>280 µg/L</u>
Chalk Creek	Nitrate-nitrogen	144 +/- 52 µg/L	<u>50 µg/L</u>	<u>24 µg/L</u>
Rock Creek	Sulfate-sulfur	20 +/- 0.66 mg/L	<u>13.13 mg/L</u>	<u>147.5 mg/L</u>
Chalk Creek	Sulfate-sulfur	27.5 +/- 14 mg/L	14 mg/L 1	8.5 mg/L

increased by nearly 400% and alkalinity more than doubled. Presumably, these changes primarily reflect a loss of hydrologic connectivity with one or more flow sources and are indicative of groundwater chemistry. Interestingly, Chalk Creek constituents responded in the opposite direction in several cases. Whereas nitrate-nitrogen increased by over 3 fold in Rock Creek, it dropped by 6 fold in Chalk Creek. Similarly, total nitrogen increased in Rock Creek but decreased in Chalk Creek.

Not surprisingly, given the substantial deviations from baseflow conditions that were observed during these events, removing measurements collected during high or low flow results in a substantial decrease in variability for many constituents. For example, when the low-flow measurements are removed from the Rock Creek dataset, the coefficient of variation for sulfate drops from 1.31 to 0.19 and that for nitrate drops from 0.75 to 0.25. Similarly, when the high flow measurements are removed from the Chalk Creek dataset, the coefficient of variation for total phosphorous drops from 2.53 to 1.23 and that for dissolved organic carbon drops from 1.18 to 0.67. This suggests that if a consistent range of flows could be defined as “base flow” at each site, it might be possible to substantially increase the statistical power to detect changes in the mean value for at least some aspects of water chemistry.

However, it should be noted that extreme events play a critical role in stream ecosystems, and that extreme values may be more important than mean values. Consequently, what may matter most is to be able to measure changes in the frequencies and/or magnitudes of these events, rather than changes in mean values. Much of the variability observed in water chemistry parameters is due to these events, meaning that a change in the variability associated with a particular measurement might serve as a surrogate for changes in the frequency and/or magnitude of extreme events even in cases where they themselves are not directly detected. This is critical, because in the absence of a large number of continuously-recording instruments collecting flow and/or water chemistry data, detecting extreme events will remain a matter of chance. Furthermore, identifying whether flow is probably higher or lower than normal at the time of a sampling visit requires either a number of years of experience at a given site, or accurate knowledge of recent events likely to impact flow (snowpack, recent precipitation history, etc.). Unfortunately, the spatial resolution of the available climate data is too coarse to be of much use in the majority of cases.

Macroinvertebrates

A preliminary analysis of patterns in macroinvertebrate community composition over the four-year sampling period was also conducted. Briefly, abundance data were log transformed to reduce the effects of extreme variations, and a Bray-Curtis dissimilarity matrix was calculated. The matrix itself was examined to assess year-to-year variation in composition at sites with multiple years of data (Table 11); in addition, cluster analysis (Figure 5) and non-metric multidimensional scaling (NMDS; Figures 6 and 7) ordination were used to look at overall patterns among sites and across years. The site with the most invertebrate samples is Chalk Creek, where samples have been collected in all four years, including both summer and fall samples. Furthermore, in 2006, five replicate macroinvertebrate samples were collected at Chalk Creek in a single day. The mean between-sample dissimilarity for these five samples was 0.19, meaning that, on average, any two replicates shared 81% of their taxa in common. In August, 2009 a similar procedure was conducted at Little Stony Creek in DENA. Amongst the four replicate samples collected there, mean Bray-Curtis dissimilarity was 0.18. Assuming that these two streams are fairly representative, this can be interpreted to mean that a combination of spatial heterogeneity in the distribution of macroinvertebrates and subsampling error (see *e.g*. Cao *et al*. 2002, Ostermiller and Hawkins 2004) limits the similarity between any two samples to approximately 80%; in other words, the sampling error averages just less than 20%.

Mean dissimilarity among all Chalk Creek samples was 0.32; however, samples were collected in both summer and fall so this figure includes seasonal as well as interannual variability and variability due to sampling error. Dissimilarity between summer-only samples was 0.32, while the dissimilarity among fall-only samples was 0.25. Given the sampling error of about 0.20, this suggests that variation in community composition is actually rather low from year to year. Similar results were obtained at Little Stony Creek (Table 11). Furthermore, in 2009 summer samples were collected from Chalk Creek in both July and August. The dissimilarity between these samples was 0.17, or about the same as the average sampling error, suggesting that community composition may not have changed at all over that time. Although it is clearly risky to draw broad conclusions based on only 4 years of data at only 2 sites, these results suggest that a reduction in sampling error could have a substantial impact on the power to detect changes in species composition.

Among all 21 streams for which summer samples were collected in multiple years (and excluding the East Fork Toklat River which is an outlier due to its depauperate invertebrate fauna) the mean between-year dissimilarity (for summer samples) was 0.36. Year-to-year dissimilarity among fall samples was also relatively low at 0.39. There was a substantial degree of variation among streams, however (Table 11), with interannual dissimilarities ranging from a low of 0.27 (Tattler Creek in DENA) to a high of 0.55 (Willow Creek in WRST). Although only a limited number of fall samples have been collected, it appears that seasonal variation (mean within-year seasonal dissimilarity = 0.46) is somewhat higher than interannual variation among samples collected in the same season. This clearly illustrates the important effect of sampling date on measured community composition. The monitoring program is continuing to expand the number of streams that are sampled in multiple seasons to better assess the relative magnitudes of interannual and seasonal variation in community composition.

Table 11. Year-to-year variation in macroinvertebrate community composition. Figures given are Bray-Curtis dissimilarities and are averages among all year pairs (*e.g.*, 2006-2007, 2006-2008, 2006-2009, 2007-2008, 2007-2009, 2008-2009 if data were available for all 4 years). The last column represents the mean within-year dissimilarity between summer and fall samples.

Stream name (n)	summer/summer	fall/fall	all samples	summer/fall(within year)
Chalk Creek (8)	0.32	0.25	0.32	0.34
Rock Creek (6)	0.36	0.39	0.41	0.36
E.F. Toklat River (5)	0.78	0.54	0.68	0.67
Jack Creek (5)	--	0.40	0.43	0.46
Little Stony Creek (4)	0.29	--	0.33	0.26
Moose Creek at bridge (3)	0.35	--	0.35	--
Little Jack Creek (3)	0.38	--	0.60	--
Willow Creek (3)	0.55	--	0.60	0.65
Igloo Creek (3)	0.47	--	0.47	--
Hogan Creek (3)	0.35	--	0.35	--
E.F. Toklat tributary (3)	0.50	--	0.50	--
Savage River (2)	0.29	--	0.29	--
Beaver Creek (2)	0.36	--	0.36	--
Lake Creek (2)	0.41	--	0.41	--
Sanctuary River (2)	0.35	--	0.35	--
Gilahina River (2)	0.32	--	0.32	--
Gorge Creek Spring (2)	0.35	--	0.35	--
McKinley Bar spring (2)	0.37	--	0.37	--
McKinley Bar Trail creek (2)	0.34	--	0.34	--
Upper Igloo Creek (2)	0.42	--	0.42	--
Tattler Creek (2)	0.27	--	0.27	--

Cluster analysis (Figure 5) provides another and in some ways more informative way to evaluate the dissimilarity matrix, because dissimilarities are considered in a relative context. Flexible-β UPGMA (Unweighted Pair Group Method with Arithmetic mean) clustering is an agglomerative hierarchical clustering algorithm, and has a number of recommended properties as an analytical method for evaluating community data (McCune and Grace 2002). For this analysis, 112 macroinvertebrate samples were included. One of the 5 June 2006 Chalk Creek replicate samples was chosen at random to represent that site visit; similarly, one of the 4 August 2009 Little Stony Creek samples was randomly selected. In most cases, samples from a given site collected in different years cluster together, even if the dendrogram is trimmed with most of the information

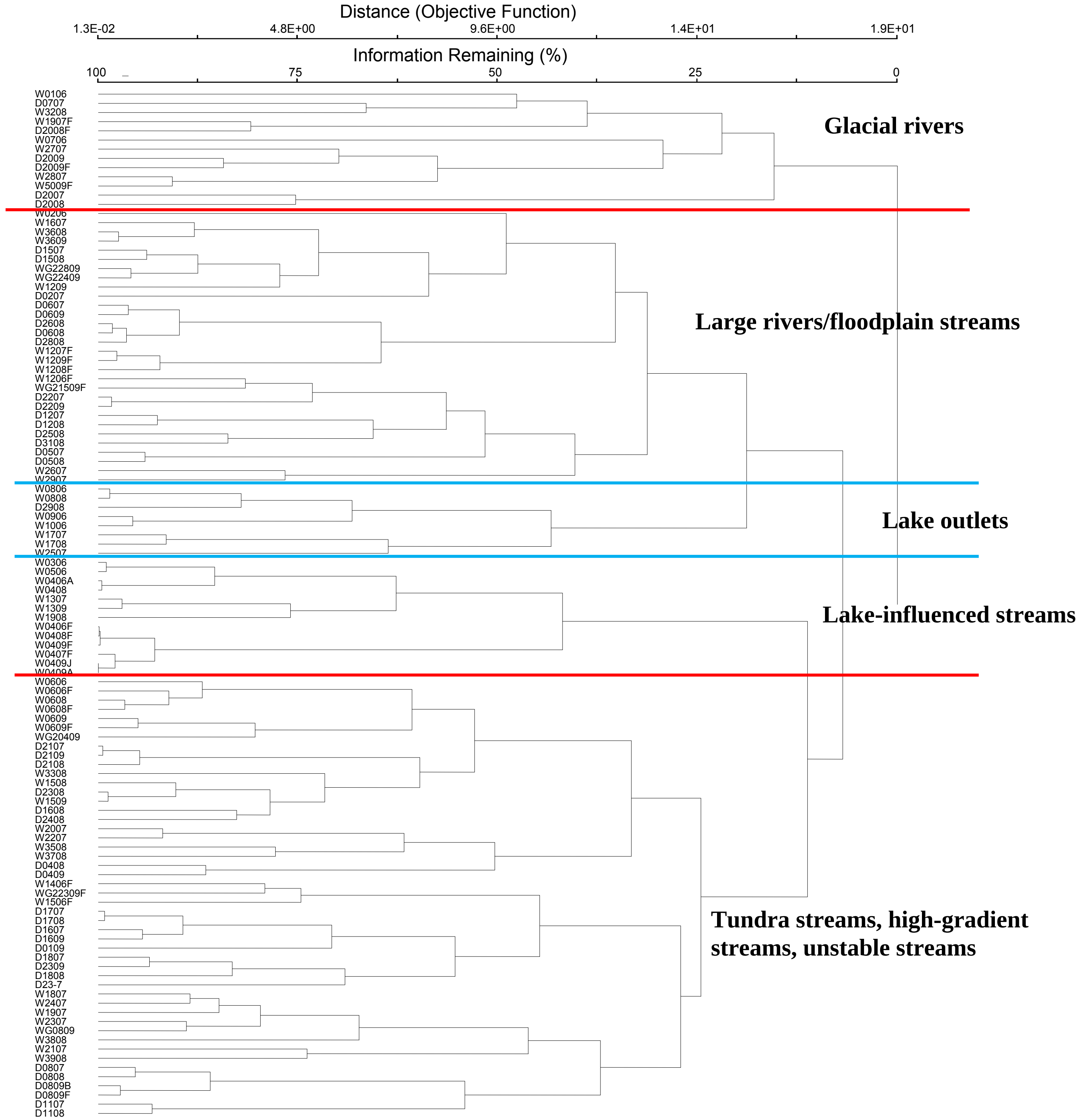

Figure 5. Flexible-β UPGMA cluster analysis (β = -0.25) of all invertebrate data from 2006-2009. Abundances were log transformed and rare taxa (occurring in < 5% of samples) were ignored. Bray-Curtis dissimilarity was used as the distance metric. Site codes (W01, D05, etc.) are listed in the Appendix. The sampling year follows the sitecode (*e.g.*, W0106), and an "F" is appended if it was a September/October sample of that site for that year (a fall sampling event).

remaining. At 75% information remaining, which is a very stringent cutoff, the majority of repeat summer samples from a given stream clustered together. At a 50% cutoff, nearly all did, including both summer and fall samples. Some of the exceptions are instructive. The failure of the Chalk Creek samples from 2006 and 2008 (W0406A and W0408) to cluster with the other Chalk Creek samples despite moderate mean dissimilarities with those samples (0.37 and 0.36, respectively) is indicative of one of the potential shortcomings of agglomerative clustering techniques, in that misclassifications can occur because later group fusions depend on earlier fusions (McCune and Grace 2002). In practice such misclassifications can be corrected, either manually or through discriminant analysis. In other instances, samples from consecutive years ended up quite "far" from each other in the dendrogram; that is, on opposite sides of a major split that occurs near the very top of the dendrogram. In some of these cases there is a reasonable environmental explanation, though no way of confirming it. Disturbance may explain the observed difference between 2007, 2008 and 2009 in the E.F. Toklat tributary samples (D2307, D2308 and D2309). The 2007 and 2009 samples are very similar and cluster together, whereas the 2008 sample is quite different. This stream is subject to aufeis events of varying magnitude every spring; substantial aufeis events are known to constitute a significant ecological disturbance and can lead to channel evulsions of varying duration. A similar pattern was observed in the Igloo Creek samples (D1607, D1608, D1609). Igloo Creek and the E.F. Toklat tributary are in adjacent basins and Igloo Creek is also subject to aufeis. Although variation in the intensity and duration of aufeis is an attractive explanation, we do not have the data to confirm or reject it.

The cluster dendrogram also reveals some broad ecological patterns in the distribution of stream biota that reflect underlying physical differences among streams. One important note is that there is little or no underlying geographical pattern; that is, the clusters reflect ecological similarities rather than physical proximity. This is best illustrated by considering lake outlet streams. Lake Creek, the outlet stream of Wonder Lake, is in DENA, but is most closely related in terms of species composition to 3 lake outlet streams in WRST - Beaver Creek and the Rock Lake outlet stream, which are 400 miles away on the Canadian border, and a tributary of the Nizina River, also called Lake Creek, located near McCarthy. In contrast, it is only distantly related to Moose Creek, into which it flows and which is less than a mile away.

The main division in the dendrogram (the split on the far right of Figure 5 that occurs at or near 0% information remaining) reflects the substantial physical and ecological differences between highly unstable stream and river ecosystems and the majority of streams and rivers that are more stable. Most of the samples above this split (labeled "glacial rivers" in Figure 5) are either from glacial rivers or streams with intermittent flow or unstable beds. Other significant divisions are illustrated in Figure 5, and all seem to be related to stability in some fashion. For example, all of the lake outlet samples from all years cluster together. Lake outlets tend to be very stable in terms of flow, temperature and water chemistry. Similarly, all of the larger rivers are found on this side. Most of the spring-fed streams and streams that appear hydrologically stable based on the prevalence of bryophytes also cluster with these streams. Broadly speaking, these results are consistent with those of Milner *et al.* (2006), who found a substantial effect of stream stability on community composition and interannual variation. Because the particular clustering algorithm used can affect the outcome of the cluster analysis, the same data were used in a second cluster analysis, this time employing the “group-average” clustering method. Like flexible-β, group-average clustering tends to be space-conserving (*i.e.*, the dendrogram preserves the pattern among sample units in the original data matrix) and is compatible with Bray-Curtis distance (Peck 2010). The dendrogram resulting from group-average clustering was similar in structure to the dendrogram depicted in Figure 5 (not shown).

Non-metric multidimensional scaling (NMDS) is a free ordination technique that is particularly suited to analysis of ecological data, which often fail to meet the assumptions of other multivariate analytical methods (Peck 2010). An NMDS ordination of the CAKN invertebrate data from 2006-2009 is shown in Figures 6 and 7. The best solution was 3-dimensional (stress was less than 20 and significantly less than a solution with randomized data); however, only the first two dimensions are depicted. The ordination has an r^2 of 0.79, which indicates that much of the variation in the original data matrix is captured accurately. As was the case for cluster analysis, no geographic pattern is apparent (compare the distribution of red [WRST] and green

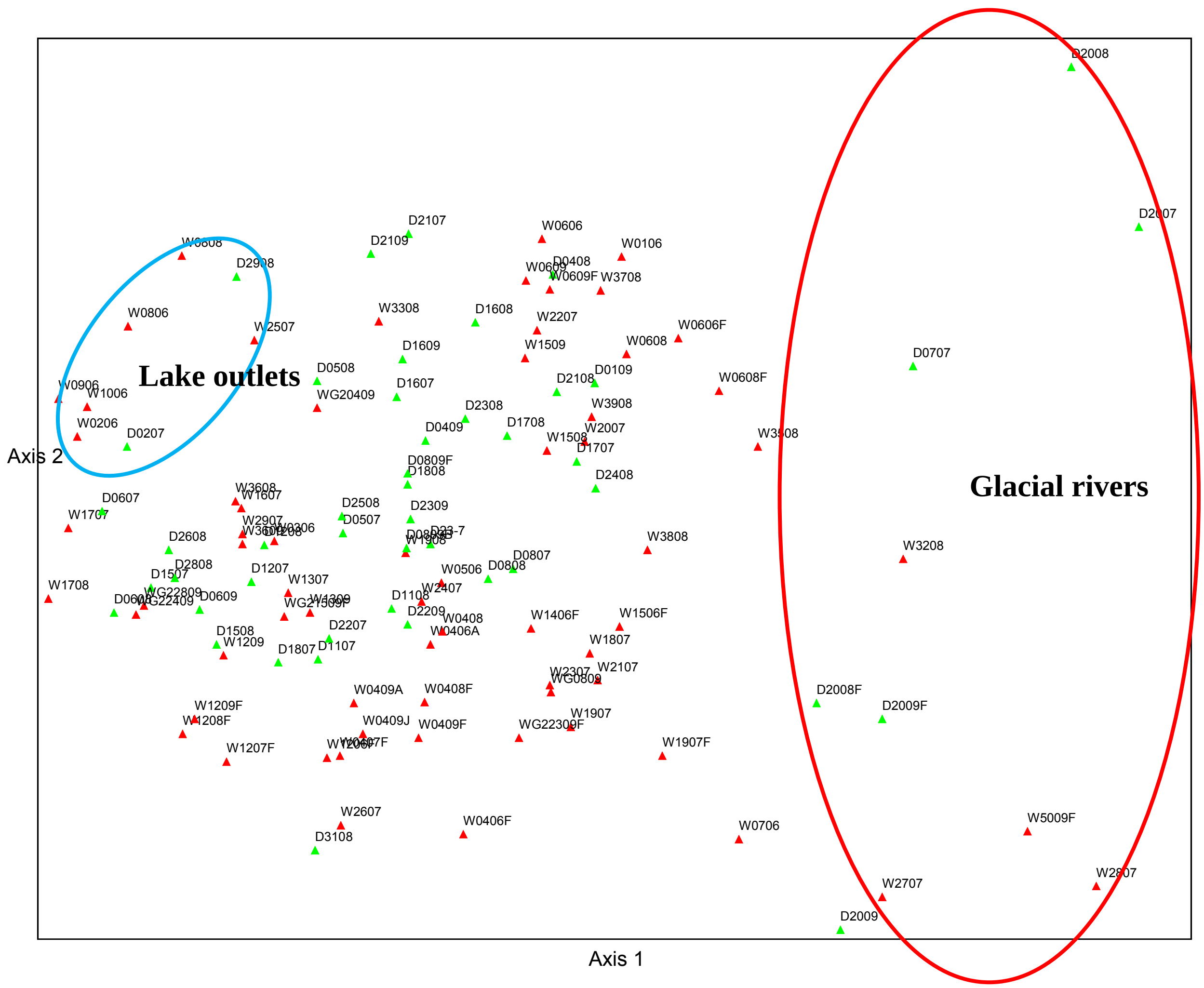

Figure 6. NMDS ordination of invertebrate data from 2006-2009. Abundance data were log transformed to reduce the effects of extreme values. Bray-Curtis dissimilarity was used as the distance measure. Site/year codes are the same as in Figure 5 and are listed in the appendix. DENA sites are shown in green, WRST sites in red. Axes 1 and 2 are shown.

[DENA] symbols). This suggests that differences in community composition among these sites are largely driven by local-scale environmental filters (*sensu* Poff 1997). Ecologically similar sites tend to be grouped in the ordination. Two of these groups are depicted in Figure 6. Lake outlets (stable) and glacial rivers (unstable) are grouped on opposite ends of Axis 1, suggesting that this axis is correlated with some measure of ecosystem stability. Note that although all of the samples from glacial and unstable streams are found together at the right end of Axis 1, the ordination positions of these samples are distributed along the entire length of Axis 2. This is because although the community compositions at these sites are distinct from those at the majority of other sites, they are quite different from each other as well. In addition to providing insight into ecological patterns among sites (spatial variation), NMDS can also be used to assess temporal variation and trends in community composition at individual sites. In Figure 7, all of the samples collected between 2006 and 2009 at each of three sites (Rock Creek, Little Stony Creek and Chalk Creek) are circled. It can be seen that the variation between samples collected at any given site (temporal variation) is substantially less than variation between sites (spatial variation), consistent with what other analyses have indicated. Successional vector analysis was conducted for sites with at least three years of data. This method can reveal whether year-to-year

Figure 7. NMDS ordination of invertebrate data from 2006-2009 showing year-to-year variation at 3 sites. This shows the same ordination depicted in Figure 6, but with only samples from 3 streams labeled.

changes in composition are directional or locally random. Directional changes in ordination space may indicate an ongoing shift in composition, *i.e*., a trend. Directional change in ordination space was apparent at one site, Little Stony Creek; however, with only three years of data it is not possible to conclude that a directional shift in community composition is occurring. As more data are collected, these methods will become more useful in assessing both spatial and temporal patterns in the distribution of invertebrates.

Development of RIVPACS biological assessment models

Accurate assessment of ecological condition (or status) at any given stream is challenging. This has become the focus of a major international research effort over the last 25 years. Detecting trends in ecological condition, either at individual sites, or across defined areas (average condition) is similarly difficult. Quantifying status or detecting trends requires that natural variation, both among and within sites, be minimized, yet commonly used summary statistics obscure these differences. Ideally, any methodology used to address these challenges will be both site-specific and standardized. Site-specific assessments of ecological condition minimize both false positives (detecting differences or trends where none exist) and false negatives (failing to detect differences or trends that do exist). Standardized assessments allow meaningful comparisons to be made among disparate sites. Such assessments have two roles in the development and implementation of the CAKN stream monitoring program. First, detecting changes in otherwise pristine stream ecosystems in response to large-scale drivers such as climate change; and second, assessing the effects of local anthropogenic drivers (e.g., road construction, resource extraction) on the ecological condition of potentially affected stream ecosystems. In either case, it will be desirable to be able to apply assessments at the level of individual streams as well as at larger spatial scales.

One such approach to ecological assessment is known as “RIVPACS-type” modeling, after the original RIVPACS model developed in Great Britain in the 1980s (see *e.g*. Clarke *et al.* 2002). Essentially, RIVPACS models use natural environmental gradients to predict the species composition expected to occur at a site in the absence of impairment. Hence these models account for much of the natural environmental variation among sites. The idea is that changes in the expected species composition constitute a metric of ecosystem impairment (or change). In other words, these methods rely on the idea that species responses are indicators of ecosystem condition (see, *e.g*. Hawkins 2006). Originally designed for macroinvertebrate assemblages, they have now been applied to a variety of other assemblages, including diatoms (Carlisle *et al*. 2008), amphibians, (Knapp *et al.* 2005) zooplankton (Knapp *et al*. 2005) and fish (Hawkins 2006). Because macroinvertebrates are ubiquitous, fill a variety of ecological niches, are sensitive to environmental stress, and are relatively inexpensive to collect and analyze, the vast majority of bioassessment programs worldwide still rely on these organisms as indicators of water quality.

In essence, a RIVPACS model predicts how many native taxa would be expected at a site in the absence of impairment. Briefly, the methodology is as follows. The distribution of macroinvertebrate taxa across a set of unimpaired (reference) sites is used in a cluster analysis to divide the streams into groups. Discriminant analysis is then used to identify sets of environmental variables that best predict the groupings. The discriminant procedure (either linear discriminant analysis or random forests) is then used to predict the probability that each reference stream is a member of each group. These probabilities of group membership for each site are then combined with the frequency of occurrence of each taxon in each group to derive probabilities of capture for each taxon at each site. Summing these probabilities gives the expected number of native taxa (E) that would be expected to be captured at each site in the absence of impairment. The actual number of expected taxa that are captured (O) is then compared to E. The resulting ratio (O/E) represents the degree to which the native fauna is intact. For reference sites, this ratio should be 1.0; that is, all of the expected taxa should be present. Substantial deviations away from 1.0 represent loss (or gain) of native taxa. For example, an O/E ratio of 0.5 would indicate that 50% of native taxa have been lost, and would suggest significant impairment.

In collaboration with Jeff Ostermiller, a bioassessment consultant, the CAKN has begun to explore the utility of these methods for detecting change in network stream ecosystems. Using macroinvertebrate data (131 taxa) from 65 calibration sites in DENA and WRST, we recently began developing a RIVPACS model for the CAKN. We also built a null model (Van Sickle *et al*. 2005), which ignores environmental gradients, to test the improvement provided by the model. Finally, we built a modification of the RIVPACS model which uses an index of Bray-Curtis dissimilarity rather than taxa lists to quantify change in the macroinvertebrate assemblage (Van Sickle 2008). This modification has several potentially advantageous features, including an increased sensitivity to changes in composition that are not accompanied by changes in richness, and the inclusion of rare taxa (which often decrease the sensitivity of traditional RIVPACS models and so are excluded).

Table 12. List of globally important predictor variables from random forest discriminant analysis of CAKN macroinvertebrate data. Climate variables are derived from PRISM climate models. Except as indicated, all variables refer to the drainage basin rather than the sampling reach. Listing is in order of importance value.

Variable name	Meaning
AnnMinTp	Annual minimum temperature
AnnMxTemp	Annual maximum temperature
JulyMxTemp	Mean July maximum temperature
Lake	% Of basin area classified as lake
Lightmodmoraine	% Of basin classified as lightly modified glacial moraine & drift
MinElev	Minimum basin elevation
OctMxTemp	Mean October maximum temp
ReachLatitude	Latitude of sampling reach
ReachLongitude	Longitude of sampling reach
SDAPRMinTp	Standard deviation of April minimum temperature (variability)

The initial results of this effort have been promising. Cluster analysis (flexible-β UPGMA) was used to separate the 65 sites into 7 groups. Random forest discriminant analysis was applied, and used to calculate posterior probabilities of group membership for each calibration site. Ideally this procedure would be repeated with a set of validation sites, but at present we do not have sufficient data. Nevertheless, calculating the O/E scores for the calibration sites provides a measure of both precision and bias in the model. Table 12 lists the globally important predictor variables from the random forest discriminant analysis. The majority of these variables relate either directly or indirectly to climate, which is consistent with what we would expect for Alaska. One important note is that in a typical discriminant analysis, most of the predictor variables that are important for predicting individual groups are also shared across groups, so that there is substantial overlap with the set of globally important variables (most important for all groups). However, for the CAKN data, this is not the case. This is most likely due to the very high degree of physical and biological heterogeneity among the CAKN streams sampled to date, and to the relatively small number of sites used for the modeling.

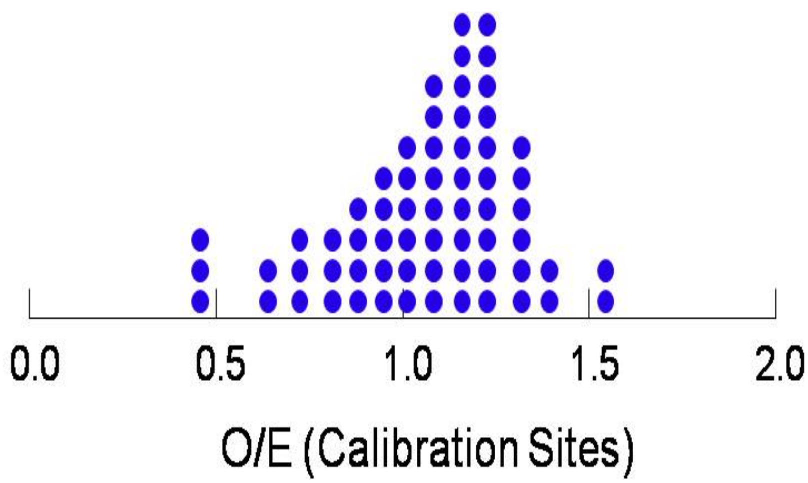

Figure 8. Distribution of O/E scores for calibration sites. Each dot represents a single site.

Figure 8 shows the distribution of O/E scores for the calibration data set, obtained by running the calibration site data through the finished model. The mean O/E score was 1.07, indicating a slight bias to the model in that on average it predicts more taxa than were actually observed. The precision of the model, as represented by the standard deviation of reference site scores, was 0.24, which although high, is comparable to other large-scale models and appreciably better than the precision of the null model (0.33). There are several significant outliers, both above and below 1.0, all of which are glacial streams and rivers. These systems are naturally depauperate and hence will need to be treated separately. Their removal from the dataset should substantially improve the performance of subsequent models. Another factor affecting the precision of the model is the combination of very high heterogeneity and a relatively small number of sites. The dataset contains 131 unique taxa, many of which are found at only a few sites, and 2/3 of which never had a probability of capture (P_c) above 0.5 at any site. Taxa with a P_c <0.5 (that is, relatively rare taxa) are normally removed from the model as a way of increasing model performance. This makes generating accurate predictions challenging. To increase the number of sites in the model, we have obtained additional data from both the Environmental Protection Agency (46 sites) and the U.S. Geological Survey (21 sites). Although not all are in CAKN parks, inclusion of these sites, in addition to data from another 13 network sites sampled by CAKN in 2010, should substantially improve model precision.

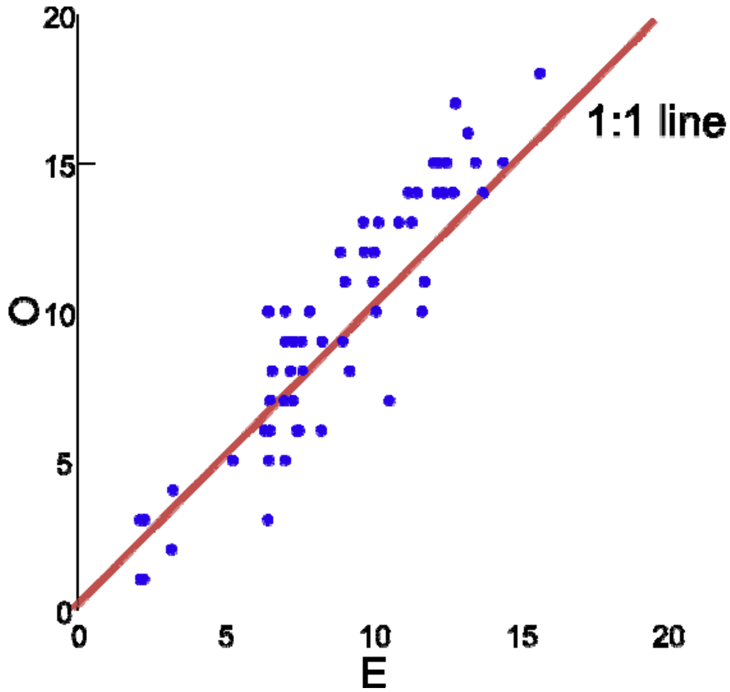

Figure 9. Plot of O vs. E for calibration sites. $R^2 = 0.85$, not significantly different from the 1:1 line.

A second measure of model performance is model bias. Figure 9 shows a plot of modeled (E) vs observed (O) invertebrate taxa at each of the 65 calibration sites. Although the values are well correlated ($r^2 = 0.85$) and the distribution is not significantly different from the 1:1 line, it can be seen that O values are consistently above the 1:1 line for all E values greater than 12. This indicates a slight model bias, particularly for more diverse streams. Once again, this is probably related to the need to incorporate data from additional sites to help mitigate the effects of the high biological heterogeneity in the dataset.

The Bray-Curtis model also appears to be promising. These models use an index (BC) derived from Bray-Curtis dissimilarity to estimate the dissimilarity between the expected and observed macroinvertebrate assemblage at a site, rather than just the number of expected taxa. In this case, reference sites are expected to have BC values near zero (indicating that the observed assemblage is nearly identical to the expected assemblage). Large BC values indicate increasing dissimilarities between observed and expected assemblages and suggest biological impairment. For the 65 sites in the CAKN dataset, the BC model was reasonably accurate and precise (mean BC = 0.25, standard deviation = 0.08). Incorporating additional data will provide a substantial improvement in the performance of these models as well.

Taken together, these results suggest that the CAKN will be able to use RIVPACS models to quantify deviations from expected community composition at individual streams, and hence to detect both impairment resulting from localized effects of anthropogenic stressors, as well as to detect changes in otherwise pristine sites resulting from remote stressors such as climate change.

In the former case, the tools we develop will allow park managers to evaluate the effects of management activity within parks, or the effects of anthropogenic stressors originating near parks, on individual stream ecosystems, as well as being able to report on the average condition of park streams. In the second case, by modeling expected assemblage composition based on a fixed temporal baseline period (*e.g.*, 2005-2010), the CAKN will be able to detect and quantify changes that have occurred in an otherwise pristine stream, even when that stream is being visited for the first time. This means that the RIVPACS approach has the potential to markedly increase the power to detect changes across the landscape, as streams do not have to be sampled repeatedly over a long time before such changes can be confidently identified.

Appendix

Translation of site codes from Figures 5-7. Sites are listed in the order shown in Figure 5, going from top to bottom of the dendrogram. Lines correspond to splits depicted using colored lines in Figure 5.

Site code	Park	Site name	Sample date	Notes
W0106	WRST	Trib West of Tana River	June 2006	Chitina River trib – intermittent?
D0707	DENA	Highway Pass Creek	August 2007	braided, unstable
W3208	WRST	Notch Creek trib	June 2008	high-gradient, unstable
W1907F	WRST	Willow Creek (Fall)	September 2007	Tana River trib
D2008F	DENA	E.F. Toklat River (Fall)	September 2008	glacial
W0706	WRST	Gravel Creek	August 2006	intermittent flow
W2707	WRST	Icy Bay stream	September 2007	young stream (glacial recession)
D2009	DENA	E.F. Toklat River	August 2009	glacial
D2009F	DENA	E.F. Toklat River (Fall)	September 2009	glacial
W2807	WRST	Independence Creek	September 2007	glacial
W5009F	WRST	Nadina River (Fall)	September 2009	glacial
D2007	DENA	E.F. Toklat River	August 2007	glacial
D2008	DENA	E.F. Toklat River	August 2008	glacial
MAJOR DIVISION IN COMMUNITY COMPOSITION				
W0206	WRST	Chakina River braid	June 2006	large river side channel
W1607	WRST	Nizina River trib	July 2007	large river floodplain stream
W3608	WRST	Rufus Creek	June 2008	source in beaver ponds, stable flow
W3609	WRST	Rufus Creek	July 2009	source in beaver ponds, stable flow
D1507	DENA	McKinley Bar Trail creek	August 2007	springfed, extensive wetlands
D1508	DENA	McKinley Bar Trail creek	August 2008	springfed, extensive wetlands
WG22809	WRST	Hunter Creek	August 2009	large river floodplain stream
WG22409	WRST	Clear Stream tributary	August 2009	large river floodplain stream
W1209	WRST	Jack Creek	July 2009	large stream, source is lake
D0207	DENA	Savage River spring creek	August 2007	large river floodplain stream
D0607	DENA	Moose Creek @bridge	August 2007	large clearwater river
D0609	DENA	Moose Creek @bridge	August 2009	large clearwater river
D2608	DENA	Moose Creek @crossing	August 2008	large clearwater river
D0608	DENA	Moose Creek @bridge	August 2008	large clearwater river
W1207F	WRST	Jack Creek (Fall)	October 2007	large stream, source is lake
W1209F	WRST	Jack Creek (Fall)	September 2009	large stream, source is lake
W1208F	WRST	Jack Creek (Fall)	September 2008	large stream, source is lake
W1206F	WRST	Jack Creek (Fall)	September 2006	large stream, source is lake
WG21509F	WRST	Nadina River trib	September 2009	small springfed stream
D2207	DENA	Savage River	August 2007	large clearwater river
D2209	DENA	Savage River	August 2009	large clearwater river
D1207	DENA	McKinley Bar spring	August 2007	large river floodplain stream
D1208	DENA	McKinley Bar spring	August 2008	large river floodplain stream
D2508	DENA	Upper Moose Creek	August 2008	large clearwater river
D3108	DENA	Cantwell Creek	August 2008	large stream some glacial influence
D0507	DENA	Sanctuary River	August 2007	large river, some glacial influence
D0508	DENA	Sanctuary River	August 2008	large river, some glacial influence
W2607	WRST	Galiano West creek	September 2007	clearwater coastal stream
W2907	WRST	Little Esker Creek	September 2007	clearwater coastal stream

Site code	Park	Site name	Sample date	Notes
W0806	WRST	Beaver Creek	August 2006	lake outlet
W0808	WRST	Beaver Creek	June 2008	lake outlet
D2908	DENA	Lake Creek	August 2008	lake outlet
W0906	WRST	Rock Lake outflow	August 2006	lake outlet
W1006	WRST	Ptarmigan Creek	August 2006	lake outlet
W1707	WRST	Lake Creek	July 2007	very low gradient, lake source
W1708	WRST	Lake Creek	July 2008	very low gradient, lake source
W2507	WRST	May Creek	September 2007	small mossy stream
W0306	WRST	trib E. of Tebay	June 2006	high gradient forest stream
W0506	WRST	Swift Creek	July 2006	clearwater stream
W0406	WRST	Chalk Creek	June 2006	clearwater stream
W0408	WRST	Chalk Creek	June 2008	clearwater stream
W1307	WRST	Gilahina River	July 2007	large high gradient forest stream
W1309	WRST	Gilahina River	July 2008	large high gradient forest stream
W1908	WRST	Willow Creek	July 2008	Tana River trib
W0406F	WRST	Chalk Creek	September 2006	clearwater stream
W0408F	WRST	Chalk Creek	September 2008	clearwater stream
W0409F	WRST	Chalk Creek	September 2009	clearwater stream
W0407F	WRST	Chalk Creek	September 2007	clearwater stream
W0409J	WRST	Chalk Creek	July 2009	clearwater stream
W0406A	WRST	Chalk Creek	August 2009	clearwater stream
W0606	WRST	Rock Creek	July 2006	small, lots of aufeis
W0606F	WRST	Rock Creek	September 2006	small, lots of aufeis
W0608	WRST	Rock Creek	August 2008	small, lots of aufeis
W0608F	WRST	Rock Creek	September 2008	small, lots of aufeis
W0609	WRST	Rock Creek	July 2009	small, lots of aufeis
W0609F	WRST	Rock Creek	September 2009	small, lots of aufeis
WG20409	WRST	Tana River trib	August 2009	large floodplain stream
D2107	DENA	Hogan Creek	August 2007	small, lots of aufeis
D2109	DENA	Hogan Creek	August 2009	small, lots of aufeis
D2108	DENA	Hogan Creek	August 2008	small, lots of aufeis
W3308	WRST	Bryan Creek	June 2008	tundra stream (high flow)
W1508	WRST	Little Jack Creek	August 2008	small clearwater stream
D2308	DENA	E.F. Toklat tributary	August 2008	small, lots of aufeis
W1509	WRST	Little Jack Creek	July 2009	small clearwater stream
D1608	DENA	Igloo Creek	August 2008	small clearwater stream
D2408	DENA	Gorge Creek	August 2008	braided, unstable
W2007	WRST	upper Skolai clearwater	July 2007	small tundra stream
W2207	WRST	4th of July Creek	July 2007	high gradient forest stream
W3508	WRST	Solo Creek tributary	June 2008	small tundra stream
W3708	WRST	N.F. White River	July 2008	small tundra stream
D0408	DENA	Travertine stream	August 2008	small tundra stream
D0409	DENA	Travertine stream	August 2009	small tundra stream
W1406F	WRST	Skookum Creek	September 2006	high gradient, unstable
WG22309F	WRST	Sheep Gulch	September 2009	high gradient tundra stream
W1506F	WRST	Little Jack Creek	September 2006	small clearwater stream
D1707	DENA	Tattler Creek	August 2007	small high gradient unstable
D1708	DENA	Tattler Creek	August 2007	small high gradient unstable
D1607	DENA	Igloo Creek	August 2007	small clearwater stream
D1609	DENA	Igloo Creek	August 2009	small clearwater stream

Site code	Park	Site name	Sample date	Notes
D0109	DENA	Rock Creek	August 2009	small high gradient stream
D1807	DENA	Igloo Creek abv Tattler	August 2007	small clearwater stream
D2309	DENA	E.F. Toklat tributary	August 2009	small, lots of aufeis
D1808	DENA	Igloo Creek abv Tattler	August 2008	small clearwater stream
D2307	DENA	E.F. Toklat tributary	August 2007	small, lots of aufeis
W1807	WRST	Young Creek	July 2007	large tundra stream
W2407	WRST	Amy Creek	July 2007	large tundra stream
W1907	WRST	Willow Creek	July 2007	Tana River trib
W2307	WRST	Bear Creek	July 2007	high gradient forest stream
WG0809	WRST	McCarthy Creek	August 2009	large stream glacial influence
W3808	WRST	Horsfeld Creek	August 2008	high gradient forest stream
W2107	WRST	Moonshine Creek	July 2007	glacial influence unstable
W3908	WRST	Caribou Creek	August 2008	intermittent flow
D0807	DENA	Little Stony Creek	August 2007	small tundra stream
D0808	DENA	Little Stony Creek	August 2008	small tundra stream
D0809	DENA	Little Stony Creek	August 2009	small tundra stream
D0809F	DENA	Little Stony Creek	September 2009	small tundra stream
D1107	Gorge Creek Spring		August 2007	small tundra spring stream
D1108	Gorge Creek Spring		August 2007	small tundra spring stream

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