



Central Alaska Network Flowing Waters Monitoring Program

2011 Annual Report

Natural Resource Report NPS/CAKN/NRR—2015/965



ON THE COVER

Aerial view of the Yukon River in Yukon-Charley Rivers National Preserve
Photograph by: Trey Simmons

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Abstract

The 2011 field season marked the sixth year of development and implementation of the flowing waters portion of the Central Alaska Network (CAKN) Vital Signs Inventory and Monitoring Program and the first year in which data were collected in all three network parks — Wrangell-St. Elias National Park and Preserve (WRST), Denali National Park and Preserve (DENA) and Yukon-Charley Rivers National Preserve (YUCH). The program goals for 2011 were to 1) continue to refine field protocols and logistics related to the collection of relevant data in network streams and rivers, 2) implement a multi-panel sampling design in WRST and DENA, 3) begin sampling GRTS-selected synoptic sites in YUCH, and 4) implement the long-term flowing water monitoring program. The data collected included biological (benthic macroinvertebrate and diatom samples), physical (channel geometry, substrate size, etc.) and chemical (collected *in situ* as well as in water samples for later analysis of chemical constituents and stable isotopes) information. Data were collected during a total of 101 site visits at 43 unique sites across the three park units. Individual sites were visited anywhere from one to five times during the open-water season, although the full suite of data was not collected at every visit. Continuously recording temperature data loggers were deployed at 24 sites in DENA and WRST. Preliminary analysis of these data reveals considerable variation in temperature regime within and among streams, but no discernable trends. It was the fourth year of active-layer depth measurements and collection of continuous soil temperature data at the two Circumpolar Active Layer Measurement (CALM) plots established in WRST.

In DENA, another seven previously unreported macroinvertebrate taxa were documented, bringing the five-year total of newly-documented taxa in that park to 50. Network-wide, the program has now documented 166 unique macroinvertebrate taxa and 460 diatom species. Of these, 16 invertebrate taxa and 20 diatom species were first documented in samples collected in 2011. The fifth year of data collected from streams involved in the Long-Term Ecological Monitoring (LTEM) program in DENA confirmed earlier indications that there are incompatibilities between data collected using LTEM methodologies and those collected using CAKN methodologies. Some preliminary analyses were conducted on the year-to-year variability of water chemistry and macroinvertebrate data collected since 2006. A substantial portion of within-site temporal variation in water chemistry appears to be attributable to differences in discharge among sampling events. Grouping sampling events by season substantially reduces this variation by partially accounting for differences in discharge, which is not typically measured at each sampling visit. 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. No directional trends in invertebrate community composition were apparent at any of the 14 streams with at least three years of data. Invertebrate and environmental data collected in 2010 and 2011 were used to refine the preliminary RIVPACS predictive biological assessment model that was first constructed for network streams using data collected from 2006 to 2009. The addition of new data improved the performance of the model, with precision and bias of the models comparable to that seen in other large-scale models.

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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, whether resulting from climate change or the effects of other natural or anthropogenic drivers. These components include hydrologic regime, geomorphology (habitat structure), temperature regime, water quality and the distribution and abundance of freshwater fish, benthic macroinvertebrate and diatom species. The overall goal is to develop a logistically feasible, repeatable and scientifically robust monitoring program that will detect change in any or all of these ecosystem indicators. 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 2011, the specific goals 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 to continue implementation of the multi-panel survey design in Wrangell-St. Elias National Park and Preserve (WRST) and in DENA. In addition, a major focus in 2011 was to begin sampling probabilistic synoptic sites in Yukon-Charley Rivers National Preserve (YUCH) with an eye toward evaluating the logistical feasibility of implementing the multi-panel survey design for the monitoring program in all three network parks. The design includes annual sampling at a moderate 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. For YUCH, the goal is to visit four sites each year on an approximately 10-year return interval, for a total of 40 synoptic sites. When fully implemented, the goal is to sample 10% of these sites (20) each year, with a total of 200 synoptic sites. The final selection of sentinel sites is ongoing. However, based on results from the first several years of the program, it is likely that ~20 sentinel sites will be in the final set, 8–10 from DENA, 8–10 from WRST, and 2–3 from YUCH. In addition to the regularly-scheduled sampling of synoptic and sentinel sites, other “judgment” sites will be sampled as circumstances allow. Aside from simply providing “more data”, judgment sites will be used specifically to increase the range of ecological gradients represented in the data set, which will improve ecological modeling efforts.

In 2011, CAKN was able to sample eight sites from the GRTS list in WRST. One of these, WRST-GRTS-130 (Jack Creek), coincides with a road-accessible sentinel site that has been sampled at least

once per year since 2006; two others are also road-accessible and have been sampled multiple times (Caribou Creek since 2009 and Crystal Creek since 2010). However, only water chemistry data have been collected to date from the latter two sites. The other five were new sites and were reached using a helicopter. An additional GRTS site, located within the perimeter of the 2009 Chakina Fire, was submerged beneath a temporary lake caused by a landslide, and could not be sampled. This brings the four-year total to 33 GRTS sites (including Jack Creek), suggesting that although the target of 10 GRTS sites per year may be feasible, at least as an average (12 unique GRTS sites were sampled in 2009, and 9 in 2010), revising the target downward may be advisable. In addition to GRTS sites, 10 sentinel sites along the road system in WRST were sampled (11 if Jack Creek is included). Table 1 provides a list of sites sampled in 2011.

In DENA, a substantial fraction (6 of 11) of sentinel sites located along the Park Road are also GRTS sites, and these have been sampled approximately annually since 2008. Three additional GRTS sites located along the road have been sampled since GRTS-based sampling began in DENA in 2010, along with four remote sites accessed by helicopter. Hence, although the total number of GRTS sites sampled in 2010–2011 (13) exceeds the target number (12), in fact only seven non-sentinel sites have been sampled, again suggesting that sampling six GRTS sites per year may not be feasible over the long term. In YUCH, five GRTS sites were sampled in 2011, the first year that GRTS-based sampling was attempted in that park unit; although this exceeds the annual target of four sites, it remains to be seen whether this rate is sustainable.

Invertebrate and environmental data collected from 2011 were combined with data from earlier years to improve the performance of the RIVPACS (or O/E) biological water quality assessment models that the program first developed using data collected from 2006 to 2009. 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 and of water quality. 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. Inclusion of these additional data substantially improved the precision of the model, with the standard deviation of derived calibration site O/E scores decreasing from 0.24 to 0.17.

Methods

Study Area

CAKN is composed of three Alaskan conservation units (WRST, DENA, and YUCH) and encompasses 21.7 million acres spread over 150,000 square miles, with 11.8 million acres of designated wilderness, all of it in DENA and WRST. YUCH contains 1.8 million acres of suitable wilderness lands (72 percent of its total area), where management is the same as in designated wilderness. Based on total area, the CAKN represents 25 percent of the land in the national park system. These parks encompass a number of physiographic and climatic regions, including 12 of Alaska's 32 ecoregions (Nowacki et al. 2002), and consequently exhibit a wide diversity of stream and river types. Elevations in the CAKN range from sea level to over 20,000 feet. Biomes range from boreal to maritime with a substantial transition zone. Most of WRST, DENA and all of YUCH have continental climates (relatively low precipitation, long extremely cold winters, and short hot summers). The south side of the Alaska Range in DENA and the coastal areas of WRST have a maritime climate (cool wet summers, relatively mild winters with extensive snowfall).

Fluvial ecosystems in the CAKN vary from large glacial rivers to small tundra streams, from boreal streams underlain by permafrost to coastal streams flowing through temperate rainforest. Much of the flowing water in the CAKN eventually drains into the Yukon River; however, the network also includes substantial portions of the Copper and Susitna River drainages and a small part of the Kuskokwim basin, as well as numerous coastal streams along the Malaspina Forelands. Although the total length of rivers and streams in the CAKN is currently unknown, estimates based on analyses of the National Hydrography Dataset place the total at approximately 28,000 miles, or about one third of all the stream miles in the National Park System (Dean Talbot, NPS, pers. comm.).

The only river in the CAKN that has been designated as a National Wild and Scenic River is the Charley River in YUCH, although many network rivers have been proposed for Wild and Scenic designation. The state of Alaska has designated no Outstanding Natural Resource Waters in these (or any other) parks. Nevertheless, most of the flowing waters in the CAKN are relatively pristine, particularly in comparison to streams and rivers in the lower 48, despite a history of placer mining in all three park units. With the exception of a relative handful of heavily impacted systems (present in each park unit), stream and river ecosystems in the CAKN are, in general, structurally and functionally intact. Nevertheless, an extensive history of mining, especially in YUCH, the Kantishna Hills district in DENA and the Chisana and McCarthy districts in WRST remains an important source of impairment to CAKN flowing waters. For example, there are over 400 abandoned mine sites in WRST. Although only two CAKN streams have been designated as impaired under Section 303(d) of the Clean Water Act, a much larger number may have some degree of impact from the effects of placer mining.

Wrangell-St. Elias National Park and Preserve

At 13.2 million acres, WRST is the largest national park in the United States. The landscape is dominated by four mountain ranges — the Alaska, Wrangell, St. Elias, and Chugach, which together create an extensively glaciated landscape containing multiple peaks over 16,000 feet, including

18,008 foot Mt. St. Elias, as well as the largest non-polar icefield in the world. WRST is also one of four contiguous conservation units spanning the U.S.-Canada border and covering approximately 32 million acres. This area, which includes Glacier Bay National Park on the U.S. side, as well as Kluane National Park and Alsek-Tatshenshini Provincial Park on the Canadian side, has been designated by the United Nations as an International World Heritage Site. Together these parks constitute the largest internationally protected area in the world. Hydrologically, WRST is dominated by the influence of the area's extensive glaciers and icefields, which cover over 25% of the park. There are six major drainages in the park — the Copper, Chitina, Bremner, White, Chisana and Nabesna rivers. Of these, the Chitina and Bremner rivers drain into the Copper, which ultimately empties into the Gulf of Alaska, whereas the White, Chisana and Nabesna rivers all drain into the Yukon River and thence to the Bering Sea. All six of these major rivers are dominated by glacial meltwater; they are turbid, highly dynamic and flow in wide, extensively braided channels.

WRST spans three climatic zones (coastal, transitional, and continental), and includes four major mountain ranges (the Wrangell Mountains, Chugach Mountains, St. Elias Mountains, and the Alaska Range). Large expanses of open, low elevation terrain occur within the Copper River basin, a relic of the huge pro-glacial Lake Ahtna, which formed behind an ice dam at the confluence of the Copper and Chitina rivers during the Pleistocene. The valley floor is now covered with braided river channels and surficial deposits mixed from alluvium and glacial outwash. Most of the rivers and streams in WRST are heavily influenced by glacier activity. Dynamics of water processes in the landscape are controlled in part by the extreme winter weather. Five different types of permafrost occur commonly throughout the park, strongly affecting surface water dynamics. Ice flows and periodic ice jams can cause brief but sometimes catastrophic flooding in low-lying areas. WRST features a diverse fish fauna, including large salmon populations that support commercial, subsistence and recreational fisheries. The Copper River and its tributaries provide critical migration corridors and spawning and rearing habitat for sockeye (*Oncorhynchus nerka*), coho (*O. kisutch*), chum (*O. keta*), and Chinook salmon (*O. tshawytscha*), as well as steelhead (*O. mykiss*). Other fish species found in WRST rivers include pink salmon (*O. gorbuscha*), Dolly Varden (*Salvelinus malma*), burbot (*Lota lota*), arctic grayling (*Thymallus arcticus*), cutthroat (*O. clarkii*), and rainbow trout (*O. mykiss*), sculpin, suckers, and whitefish.

Denali National Park and Preserve

DENA is the oldest, best-known and most-visited national park in Alaska, and has been designated by UNESCO as an International Biosphere Reserve. DENA spans four ecoregions, including substantial areas of the Tanana-Kuskokwim Lowlands and Alaska Range Transition, along with small areas of the Kuskokwim Mountains in the northwest and Cook Inlet Basin in the south. The park covers some 6 million acres, of which over 2 million are designated wilderness. Although dominated by, and best known for, the 20,320 foot high bulk of Denali (Mt. McKinley), DENA is home to a wide variety of landscapes, ranging from the flat wetlands around Lake Minchumina in the northwest to high altitude tundra in the Alaska Range to wetter and more temperate areas in the rarely visited southern part of the park. A number of large glacially-influenced rivers flow both north and south out of the Alaska Range. There are also numerous smaller clearwater streams that arise in the Alaska Range foothills, the Kantishna Hills, and in the flat northwestern part of the park. Streams

in the northwest are more likely to have high levels of dissolved organic carbon and be highly colored. DENA streams and rivers flow into three major river basins. The south side of the Alaska Range is dominated by the Susitna Basin, whereas most streams and rivers on the north side of the range flow into the Tanana River and thence into the Yukon. The far western part of the park includes a small portion of the Kuskokwim Basin. Although subsistence and recreational fisheries are less important here than in the other network parks, DENA rivers do support a variety of fish species, including Chinook, coho, sockeye and chum salmon, Dolly Varden, arctic grayling, sculpin, and whitefish.

Yukon-Charley Rivers National Preserve

YUCH flanks 130 miles of the Yukon River, which at over 2,300 miles long is the second longest river in North America (after the Mackenzie), and which drains some 324,000 square miles of Alaska and Canada. YUCH contains the entire basin of the 106-mile-long Charley River (a designated National Wild and Scenic River). Several major tributaries of the Yukon also flow through the preserve, including the Tatonduk, Nation and Kandik Rivers. YUCH contains 2.5 million acres of remote boreal forest and upland tundra and spans three ecoregions in interior Alaska, the Yukon-Tanana Uplands, the North Ogilvie Uplands, and in the far northwest a small portion of the Yukon-Old Crow Basin (Nowacki et al. 2002). The climate is typical of interior Alaska, with short, warm summers, and long, very cold winters. With the exception of the mainstem Yukon, which is turbid from glacial silt inputs upstream (originating in the White River), all of the streams and rivers in YUCH run clear. Seventeen species of fish have been reported to occur in streams and rivers in YUCH, including coho, Chinook and chum salmon (Morrow 1980, Daum 1994). Although YUCH is primarily a transit corridor for Chinook salmon headed for spawning grounds in Canada, it has recently been discovered that juvenile Chinook from Canada migrate into the preserve to rear in small tributaries (Daum and Flannery 2009).

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 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 four-digit hydrographic units that encompass WRST, DENA or YUCH, respectively, were first merged, and then clipped to the park boundaries using ArcGIS Desktop (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 over-represent stream segments likely to be 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.

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 has been 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 turn out to be 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”. For WRST, 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. Variable conditions along these river corridors makes them unreliable access routes for fixed-wing aircraft. As a result helicopters are required for access to most of these sites. For DENA, the accessibility surface included only the sites near the Park Road, designated trails and floatplane-accessible lakes. For YUCH the initial accessibility surface included 3 levels – areas within 1 mile of the Yukon River, similar corridors along the Charley, Nation, Tatonduk and Kandik Rivers and known airstrips, and lastly remote sites. Ultimately, two access categories were used: the Yukon River corridor plus sites adjacent to airstrips were defined as accessible; all other areas were defined as inaccessible.

Once the appropriate 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. For WRST, 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 likely that the majority of sites would not be sampleable. This required a very large starting list of possibilities to arrive at a set of 100 sampleable. The unequal weighting function was used to over-represent 2nd and 3rd order stream segments in the site list.

A similar approach was used in DENA. The NHD was edited to assure topological consistency and then attributed with Strahler stream order. However, in contrast to WRST, multiple flowpaths in braided streams and rivers were not removed. As was the case in WRST, the majority of stream segments are 1st and 2nd order (51% and 20% of the total, respectively). Probably due in part to the retention of multiple flowpaths, the percentage of stream segments in DENA attributed as high order was substantially greater than in WRST. In WRST, 3% of segments were 5th order, 1% were 6th order, and there were no 7th order segments (although the Copper River was excluded from the analysis). In contrast, in DENA 7.4% of segments were 5th order, 2.7% were 6th order and 0.6% were 7th order. Unequal weighting by stream order in the final site list was identical to the weighting for WRST. Stratification by ease of access was also similar, although access points were limited to the Park Road and floatplane-accessible lakes. For DENA, the algorithm was set to select a 31-site accessible stratum with a 93-site oversample, and a 31-site inaccessible stratum with a 93-site

oversample, with a goal of 60 sites in the final set. The “extra” sites had to be included to account for the presence of 7th-order segments in the DENA NHD, although these are not part of the target population. These numbers were chosen to obtain a similar site:area ratio as was established in WRST. WRST has about 9 million acres of unglaciated terrain, while DENA has about 5 million acres (55% as much).

Much the same approach was applied in 2011 to select and evaluate GRTS sites in YUCH. The NHD was edited to assure topological consistency and merged with adjacent portions of the Canadian National Hydrography Network (NHN) to generate a seamless data layer. The resulting stream network was then attributed with Strahler stream order. Percentages of high stream order segments (5th, 6th and 7th) were substantially higher in YUCH than was the case for either WRST or DENA, due to the dominance of the Yukon River and several other larger rivers (Charley, Nation, Kandik and Tatondik) in a relatively small area. Unequal weighting by stream order for site selection used the same weights as for DENA and WRST, with 2nd and 3rd order stream segments over-represented. Sites were initially stratified by access using three categories – “very accessible” sites were defined as sites within a mile of the mainstem of the Yukon River or an airstrip, “accessible” sites were defined as sites within one mile of the Nation, Kandik, Charley or Tatonduk Rivers, and all other sites were defined as “inaccessible.” Initial examination of the resulting site lists suggested the distribution was problematic; accordingly, the latter two categories were combined into a single inaccessible stratum. A 3x oversample of each stratum was also selected. A final list of 246 potential sampling sites was generated, from which a final list of 40 will be identified for inclusion in the long-term sampling program.

All 894 sites (400 for WRST, 248 for DENA, 246 for YUCH) have been or will be examined using a variety of remotely sensed and GIS data to determine whether they are part of the target population. Although all 400 sites in WRST have been evaluated using remote imagery, this process is still ongoing for DENA and YUCH. Field evaluation of the sites determined to be part of the target population is also ongoing in all three park units.

Reach Definition

Sampling reaches were defined using guidelines from the US Environmental Protection Agency (USEPA) Environmental Monitoring and Assessment Program’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) and between repeat site visits, the loss of accuracy should be 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.

Biological Sampling

Biological sampling protocols were largely adopted from the 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 (Figure 1). 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 (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. 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 or species, 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.



Figure 1. Collecting macroinvertebrates using a modified D-net (Rufus Creek, WRST).

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.

No attempt was made in 2011 to collect fish data; this vital sign is currently being re-evaluated.

Physical and Chemical Data Collection

We used an YSI QS650 sonde to collect temperature, specific conductivity, pH and dissolved oxygen data *in situ*. Data were collected in riffles or runs and generally in the thalweg at the bottom of the sampling 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 (Figure 2). 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 kept on ice until it could be frozen (generally the same day). 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 kept on ice until it could be frozen (generally the same day). 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. At some sites, samples for stable isotope analysis were also collected. The relative concentrations of stable oxygen and hydrogen isotopes in a stream water sample can be compared to those found in precipitation, groundwater and other potential contributing water sources to help define the relative importance of various hydrologic flowpaths. Stable isotope samples are being analyzed in collaboration with Jeff Welker of the Environmental and Natural Resources Institute at the University of Alaska Anchorage.



Figure 2. Filtering water chemistry samples in the field (Independence Creek, WRST).

Physical data collection protocols were largely based on EMAP WSA protocols (USEPA 2004). From 2006 to 2008, physical habitat data were collected at every site. 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.

However, since 2009, the program has generally lacked the field personnel for physical habitat data collection, which requires a minimum of two people. Even when personnel are available, generally for helicopter-based sampling, the logistics of sharing helicopter time with other projects has necessitated substantially shortening the time spent at each site. Collection of physical habitat data can add up to two hours to the time spent at a site. Accordingly, the collection of physical habitat data is now restricted to sentinel sites, which are road accessible, and occurs only when adequate field assistance can be obtained. The current goal is to assess physical habitat characteristics approximately once every five years, and only at sentinel sites.

Continuous Temperature Data Collection

Continuously recording dataloggers have been deployed since 2008 to monitor the temperature regimes of CAKN streams. To date the Stream Monitoring Program has relied on Onset Computer Hobotemp Pro V2 dataloggers, which are relatively inexpensive, small, and reliable, and feature a five-year deployed battery life. In 2008–2009 loggers were only deployed during the open water season (May–Sept). However, starting in 2010 the loggers have remained in place throughout the year. Temperature and battery voltage are collected hourly. This interval allows the logger to collect approximately three years of data before the memory capacity runs out. Loggers are normally replaced every two years; however, the three-year storage capacity means that loggers can be left in place for an additional year if necessary. Early on in the monitoring program, a number of deployment methods were evaluated. The deployment method needed to meet two opposing requirements – it needed to be robust enough that loggers remained submerged and were not washed away by ice scour or during high flows, yet the system needed to cause minimal visual disruption to be compatible with park compliance mandates. The current deployment method is illustrated in Figure 3. The loggers are placed inside protective white vinyl cases, and attached to a vinyl-coated steel cable using a copper cable sleeve and wire rope clip. The logger is then securely attached to a large flattish rock using 18–20-gauge steel wire. The other end of the cable is then secured to a riparian tree or shrub, or in some cases to a bridge piling.



Figure 3. Deployment of temperature dataloggers. The image on the left shows the attachment of the logger to the anchor rock with 20 gauge wire, and to the vinyl-coated steel cable with a cable sleeve and wire clip. The image on the right shows a deployed logger anchored to a riparian tree, which is the preferred method.

Where possible, loggers are deployed in relatively deep, fast-flowing water (e.g., in a run). The rock is placed in the stream with the logger underneath it and additional rocks are generally placed to provide additional anchoring. Care is taken to minimize the visibility of both the logger and the anchoring system. This is important both to minimize the visual impact of data collection on park visitors, as well as to discourage vandalism and/or theft of the units.

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, including cluster analysis, ordination and MRPP, 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 USEPA and other institutions (J. Ostermiller, personal communication). Basin descriptor variables used as environmental predictors for random forests discriminant analysis were derived from GIS using data layers compiled by the National Park Service and/or ABR, Inc. Basins (defined as the upstream contributing area for each sample point) were first delineated in ArcGIS using customized scripts. Individual analytical methods are discussed in the Results and Discussion section as appropriate.

Results and Discussion

Overview of 2011 Data Collection Efforts

The study area in 2011 included all three park units in the network for the first time. As has been the case in previous years, most of the sampled sites (33/43) were located along roads. Data were collected during a total of 103 site visits at 43 unique sites in 2011. In DENA there were 43 site visits to 18 unique sites, in WRST there were 55 visits to 20 unique sites, and in YUCH there were 5 site visits to 5 unique sites (Table 1, Figure 4). The full suite of data was not collected during all visits, however. In general, invertebrates and diatoms were only collected during one visit per year, generally in mid-summer. The exceptions in 2011 were Chalk Creek and Rock Creek in WRST, where invertebrate and diatom samples were also collected during a September visit. The locations of the sites sampled are shown in Figure 4, along with the locations of all sites sampled in previous years. All of the DENA sites sampled in 2011 were located along or near the park road. These sites included 11 sites that have been sampled since 1994 as part of the LTEM program. Sixteen of the 18 sites sampled in DENA have been previously sampled by the Stream Monitoring Program at least once during 2007–2010. No remote GRTS sites were sampled in DENA in 2011 due to logistical problems.

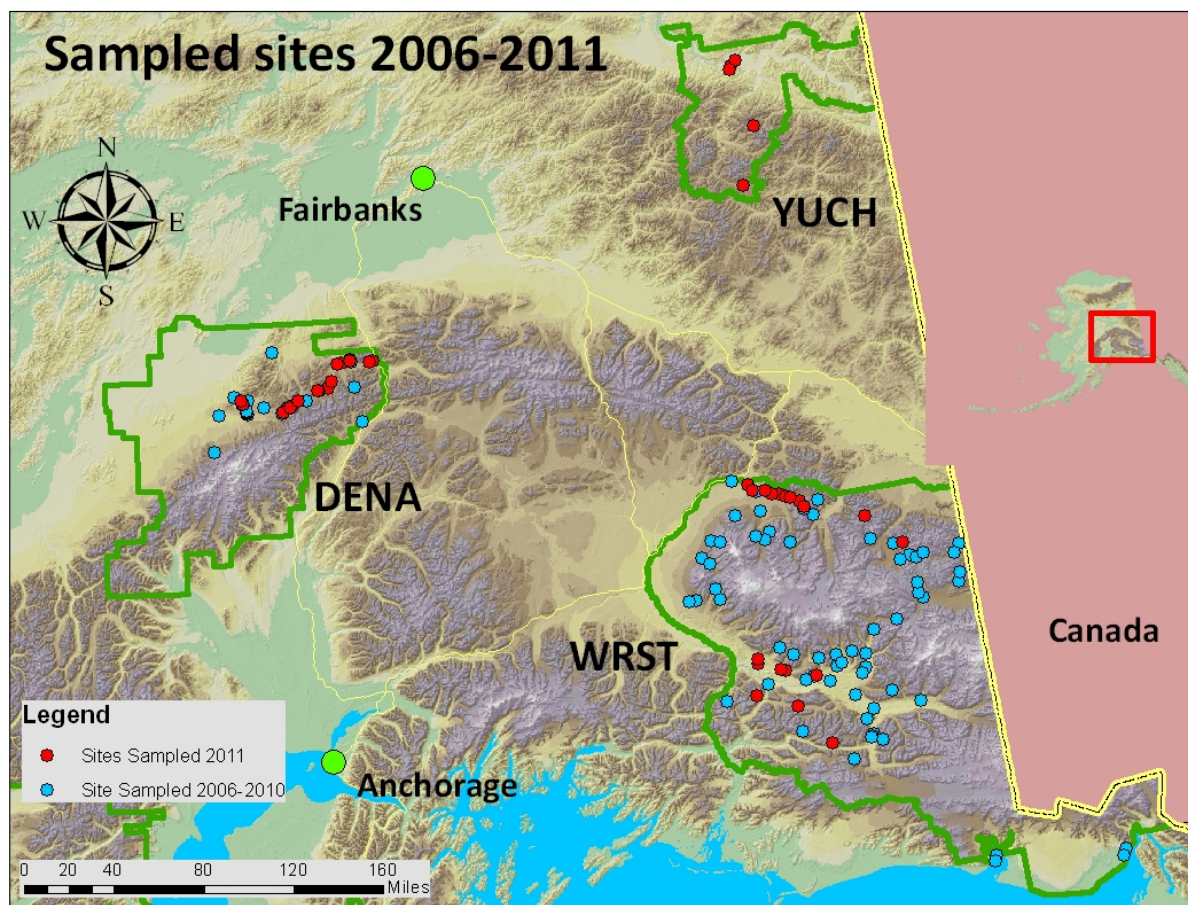


Figure 4. Locations of 2011 CAKN stream monitoring sites (shown in red). Sites sampled in previous years are shown in blue.

In WRST, 15 sites along the road system were sampled, 14 of which had been sampled in previous years. The new road site, Strelna Creek, was from the GRTS list. Four remote sites in WRST, all of them from the GRTS list, were also sampled by helicopter in 2011. All five of the sites sampled in YUCH in 2011 were from the GRTS list. Two of these sites are located along Coal Creek, and were accessed by ATV. The other three were remote and were accessed by helicopter.

Table 1. List of sites sampled in 2011. A * indicates sites that have been sampled at least once in previous years. A *b* indicates sites where biological data were collected in 2011.

Site	Park	Description
Rock Creek ^{*b}	DENA	high-gradient, confined-channel step-pool stream (LTEM site)
Travertine Stream [*]	DENA	Unusual water chemistry
Little Stony Creek [*]	DENA	Spring-fed tundra stream (LTEM site)
Gorge Creek Spring [*]	DENA	Spring-fed stream sampled at source
Tattler Creek ^{*b}	DENA	small steep braided stream (LTEM site)
Hogan Creek [*]	DENA	small groundwater-fed stream (LTEM site)
Lake Creek ^{*b}	DENA	Wonder Lake outlet stream
Hines Creek trib ^b	DENA	Similar to Rock Creek
Wonder Lake inflow ^b	DENA	Low-gradient meadow stream
Stony Creek tributary ^{*b}	DENA	very small stream on Highway Pass
Moose Creek at bridge ^{*b}	DENA	large plane bed stream in Kantishna Hills (LTEM site)
E.F. Toklat tributary [*]	DENA	clearwater tributary to E.F. Toklat River (LTEM site)
E.F. Toklat River [*]	DENA	large turbid braided glacial river (LTEM site)
Savage River [*]	DENA	large river with some glacial influence (LTEM site)
Stony Creek [*]	DENA	steep step-pool unconfined channel stream
Igloo Creek ^{*b}	DENA	large forested stream (LTEM site)
Sanctuary River [*]	DENA	large glacially-influenced river (LTEM site)
Highway Pass Creek ^{*b}	DENA	unconsolidated braided stream (LTEM site)
Chalk Creek ^{*b}	WRST	clearwater lake outlet stream along Nabesna Road
Rock Creek ^{*b}	WRST	clearwater stream along Nabesna Road
Rock Creek tributary [*]	WRST	very small DOC-rich forest stream
Gilahina River [*]	WRST	large clearwater stream along the McCarthy Road
Little Jack Creek ^{*b}	WRST	clearwater stream along Nabesna Road
Rufus Creek [*]	WRST	stable groundwater stream along Nabesna Road
Caribou Creek @ culvert [*]	WRST	Intermittent boreal stream

Table 1. List of sites sampled in 2011. A * indicates sites that have been sampled at least once in previous years. A *b* indicates sites where biological data were collected in 2011 (continued).

Site	Park	Description
Chokosna River	WRST	Moderately-sized clearwater river
Strelna Creek	WRST	Clearwater boreal forest stream
Long Lake Creek	WRST	Long Lake outlet stream
Paco Creek	WRST	Small meadow stream
Chicken Creek	WRST	High-gradient tundra stream
Caribou Creek	WRST	Intermittent boreal stream
Strelna Creek @ crossing	WRST	Clearwater boreal forest stream
Jack Creek at bridge ^{*b}	WRST	large lake outlet stream along Nabesna Road
Tanada Creek @ weir ^b	WRST	Large lake outlet
Stuver Creek tributary ^b	WRST	Small high-gradient tundra stream
Paco's River ^b	WRST	High-gradient Chugach tundra stream
Hanagita River ^b	WRST	Large pool-riffle lake outlet
Bridge Creek ^b	WRST	Large lake outlet
Upper Charley River ^b	YUCH	Larger tundra stream
Coal Creek ^b	YUCH	Substantially disturbed by historic mining
Andrew Creek ^b	YUCH	Very low gradient silt-bottomed Yukon tributary
Coal Creek @strip ^b	YUCH	Substantially disturbed by historic mining
Seventymile River trib ^b	YUCH	High gradient alpine tundra stream

Temperature-recording dataloggers were installed or replaced in 13 streams in May and early June, bringing the total number of deployed dataloggers to 24 (Table 2). In addition, two continuously recording pressure transducer/temperature recorders were installed at the Jack Creek site (WRST), and one pressure transducer/temperature recorder was deployed at Caribou Creek in conjunction with the temperature datalogger. Dataloggers were not replaced at several DENA sites due to excessive instream ice remaining at the time of the spring deployment trip. Three other temperature loggers, both installed at remote sites in WRST in 2008, could not be retrieved before the end of the 2011 field season, and remain in place, or so it is hoped. Exploratory data analysis of the first four years of continuous temperature data revealed a wide variety of temperature regimes among streams in the network. Temperature regimes also varied across years at some of the sites. Insufficient data have been collected so far to allow meaningful analyses for trend to be conducted, and exploratory analysis revealed no consistent patterns across the limited number of years available.

Table 2. CAKN sites where temperature loggers were deployed in 2011.

Site	Park	Description
Tattler Creek	DENA	Small steep braided stream (LTEM site)
EF Toklat tributary	DENA	Medium-sized alpine stream (LTEM site)
Stony Creek tributary	DENA	Very small tundra stream, intermittent
Little Stony Creek	DENA	High-elevation tundra stream (LTEM site)
Gorge Creek Spring	DENA	Alpine spring outlet
Lake Creek	DENA	Outlet of Wonder Lake
Moose Creek @ bridge	DENA	Large clearwater stream in Kantishna Hills (LTEM site)
Sanctuary River	DENA	Large low-gradient river, some glacial influence (LTEM site)
Hogan Creek	DENA	Small, stable high gradient stream
Savage River	DENA	Medium-sized braided floodplain river
Rock Creek DENA	DENA	Small high-gradient forest stream
Crystal Creek	WRST	Small steep forested lake outlet with beaver dams
Long Lake Creek	WRST	Low-gradient forested lake outlet stream
Gilahina River	WRST	Large clearwater stream along the McCarthy Road
Chalk Creek	WRST	Clearwater stream along Nabesna Road
Rock Creek	WRST	Clearwater stream along Nabesna Road
Jack Creek at bridge	WRST	Large lake outlet stream along Nabesna Road
Paco Creek	WRST	Small meadow stream
Little Jack Creek	WRST	Clearwater 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
Tana River tributary (2 nd location)	WRST	Wide plane-bed stream in Tana River floodplain

Implementation of the GRTS Survey Design

In 2008, the feasibility of using GRTS to select a probabilistic sample of synoptic sites was first tested in WRST. After editing the NHD 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. The 305 sites that remained on the list after this step continue to be evaluated in the field, primarily by helicopter overflight. The results of this field evaluation have been used to select potential sampling sites for the 2009–2011 field seasons. In general, approximately 40% of sites that are evaluated meet the criteria for sampling (a stream is actually present, a safe landing zone is nearby, stream conditions appear to allow safe sampling). For the most part, the field evaluations have held up when site visits are actually attempted. A similar approach has been taken since 2010 for selecting and evaluating GRTS sites in DENA and in 2011 for sites in YUCH.

The logistical challenges of working in remote Alaskan parklands have forced CAKN to reconsider strict application of the GRTS survey design. 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 comparable site (same accessibility stratum and stream order) on the list is sampled. However, this approach turns out to be problematic for the CAKN, where consecutive sites may be hundreds of miles apart, and will usually 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. 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. Similar challenges apply for GRTS-based sampling in DENA, where even the accessible stratum is largely derived from lake-proximal sites in the remote northwest part of the park, and to a lesser extent in YUCH. 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 tend to be in geographic proximity to one another, and that certain areas of the parks (e.g., coastal area and far northeast for WRST, south of the Alaska Range and west/northwest areas for DENA) are probably going to be underrepresented in the final site list, as it is more difficult to arrange shared helicopter time in those areas. This approach also means that the program will be unable to use the “canned” variance estimators and other statistical procedures that have been developed specifically for evaluating data collected using the strict interpretation of the GRTS survey design. Accordingly, the statistical analyses of these data will be more complex, although it is likely that Bayesian approaches and/or model-based inference methods will be readily applicable.

Water Chemistry

The water chemistry results from 2011 (Table 3) revealed a very wide range of conditions across the parks, which is consistent with results from previous years. The descriptive statistics derived from all of the water chemistry data collected from 2006 to 2011 are very similar to those calculated based on the 2011 data alone. In general the maximum and minimum values for a given solute vary from year to year much more than the mean and median values, as would be expected. With six 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 varied over several orders of magnitude across the CAKN sites sampled in 2011, which is consistent with what has been observed in the data collected since 2006. For example, specific conductivity varies from a low of 10 $\mu\text{S}/\text{cm}$ (Seventymile River tributary in YUCH) to a high of 1472 $\mu\text{S}/\text{cm}$ (Hines Creek tributary in DENA), a nearly 150-fold range. Sulfate is even more variable, ranging from 0.16 mg/L (Seventymile River tributary in YUCH) to 182 mg/L (Hines Creek tributary in DENA), and a greater than 1000-fold range. Water chemistry characteristics can also vary over very short spatial scales. For example, the Rock Creek and Rock Creek tributary sampling sites in WRST are less than 5 meters apart, but exhibit substantially different chemical profiles.

Table 3. Summary water chemistry statistics from 2011 samples.

Parameter	Minimum	Mean	Median	Maximum
Temperature (instantaneous)	-0.01°C	5.9°C	6.0°C	13.1°C
Specific conductance	10 $\mu\text{S}/\text{cm}$	382 $\mu\text{S}/\text{cm}$	337 $\mu\text{S}/\text{cm}$	1472 $\mu\text{S}/\text{cm}$
pH	6.82	8.03	8.09	8.58
Alkalinity (HCO_3^- -C)	8.92 mg/L	98.8 mg/L	84 mg/L	371 mg/L
Total nitrogen	30 $\mu\text{g}/\text{L}$	206 $\mu\text{g}/\text{L}$	180 $\mu\text{g}/\text{L}$	790 $\mu\text{g}/\text{L}$
Nitrate-nitrogen	<1 $\mu\text{g}/\text{L}$	80 $\mu\text{g}/\text{L}$	84.5 $\mu\text{g}/\text{L}$	324 $\mu\text{g}/\text{L}$
Dissolved organic nitrogen	0 $\mu\text{g}/\text{L}$	113 $\mu\text{g}/\text{L}$	70 $\mu\text{g}/\text{L}$	565 $\mu\text{g}/\text{L}$
Total phosphorous	<1 $\mu\text{g}/\text{L}$	14.7 $\mu\text{g}/\text{L}$	5 $\mu\text{g}/\text{L}$	190 $\mu\text{g}/\text{L}$
Phosphate-phosphorous	< 1 $\mu\text{g}/\text{L}$	4.6 $\mu\text{g}/\text{L}$ *	2 $\mu\text{g}/\text{L}$ *	35 $\mu\text{g}/\text{L}$
Sulfate-sulfur	0.16 mg/L	30.7 mg/L	15.6 mg/L	182mg/L
Dissolved organic carbon	0.31 mg/L	3.2 mg/L	1.5 mg/L	16.7 mg/L

Solute concentrations are given as micrograms per liter ($\mu\text{g}/\text{L}$) or milligrams per liter (mg/L). Specific conductivity is in units of microSiemens/centimeter. *70% of samples tested in 2011 had phosphate-P levels at or below the level of detectability (1 $\mu\text{g}/\text{L}$) reported by the Cooperative Chemical Analytical Laboratory (CCAL). I substituted values of $\frac{1}{2}$ the detection limit for calculating descriptive statistics, although with the understanding that it may introduce bias (Helsel 2006).

Another pattern that was confirmed with the 2011 data is the generally low levels of phosphate-P in CAKN streams. The mean concentration of phosphate-P among all samples collected from 2006 to 2011 was less than 3 $\mu\text{g}/\text{L}$, and 60% of samples had undetectable levels (<1 $\mu\text{g}/\text{L}$). Total phosphorus (TP) was also generally low, with a few exceptions. The mean TP concentration in CAKN streams sampled to date (28 $\mu\text{g}/\text{L}$) is three times lower than the comparable value (90 $\mu\text{g}/\text{L}$) for wadeable streams in the lower 48 (USEPA 2006). The median concentration for CAKN streams is only 9 $\mu\text{g}/\text{L}$, which is strikingly low. In fact, only a handful of streams in the CAKN had a TP concentration higher than 50 $\mu\text{g}/\text{L}$ and 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 in those streams 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”, Redfield 1958) 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 41:1, and the range goes as high as 1500:1 (Rock Creek in DENA). Over 80% of the samples collected have N:P ratios greater than 16. As expected, most of the streams with N:P ratios <16 are glacially influenced, so the high TP values are not reflective of biologically available phosphorous and these systems are also mostly P-limited. No obvious patterns emerge that explain the distribution of N:P ratios. For example, one of the lowest (Nadina River, 0.55) and one of the highest (E.F. Toklat River, 653) ratios were observed in glacial rivers. In general, however, lake outlets and stable groundwater-dominated streams had the lowest N:P ratios. 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 TP among samples. Because for most samples, TP was near the limit of detection, the variation in N:P, both within and among sites, is probably exaggerated somewhat.

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 194 samples so far analyzed, nearly 60% are below the level of detection for SRP (1 μ g/L) and 73% are below the level of detection for ammonium nitrogen (10 μ g/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 (fewer 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), although within a year or two we may have enough detected values for survival analysis.

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, CAKN began requesting TDN analysis in 2009. However, the initial 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 2010, the program switched to polycarbonate filters. Although this eliminated the contamination problem, the smaller pore size (0.40 vs 0.45 μm for nitrocellulose) has made field filtration more of a challenge, adding time to the filtration process and with multiple filters often being required. Results from the 2011 CAKN samples are in line with the results from 2010 and with expectations based on the literature, with DON comprising a substantial proportion of total TDN. Because TDN is almost always >90% of total N, DON also comprises an important fraction of the total nitrogen in these streams (Table 4).

Table 4. Characteristics of dissolved organic nitrogen (DON) in CAKN streams sampled in 2010–2011 (n = 72).

	DON Concentration ($\mu\text{g/L}$)	% of Total Dissolved N
Minimum	0	0
Mean	81	40%
Median	53	32%
Maximum	565	98%

Macroinvertebrates

A total of 25 macroinvertebrate samples were collected from 8 DENA streams, 13 WRST streams (15 site visits), and 4 YUCH streams in 2011. As was the case in previous years, observed OTU (Operational Taxonomic Unit) taxa richness, a conservative estimate of true species richness, varied tremendously among the sampled sites, ranging from 7 unique taxa at the high elevation Stuver Creek tributary in WRST to 32 at Jack Creek in WRST. Taxon richness at Jack Creek is consistently among the highest of any network stream. The mean richness across all samples collected in 2011 was 22.4 unique taxa, which is similar to the mean richness observed in previous years (mean = 20). Among 2011 invertebrate samples, chironomid midges accounted for approximately 27% of OTU taxa richness across all sites, and 28% of individuals. This is substantially lower than reported by Oswald (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 (2006–2010 mean % richness as chironomids was 32% and mean % of total individuals as chironomids was 38%). Densities varied from a low of 91 individuals per m^2 at Coal Creek in YUCH to a high of nearly 9,000 per m^2 at Lake Creek in DENA, with a mean of 2659 individuals per m^2 , also similar to what has been observed in previous years.

Sixteen taxa not previously documented by CAKN were collected in 2011 (Table 5), including a new family of aquatic snails (Physidae) and a new family of caddisflies (Hydroptilidae). Between 2007 and 2010 a total of 43 aquatic invertebrate taxa not reported by Milner et al. (2003) or by Conn (1998) were collected by the CAKN crew in DENA (Simmons 2010, 2011, 2013). In 2011 we collected an additional seven previously unreported invertebrate taxa in DENA (Table 5). This brings the five-year total to a minimum of 50 newly reported aquatic macroinvertebrate taxa in DENA,

which is remarkable given the limited geographic and temporal scope of the CAKN program in DENA and supports the idea that even after two decades of sampling, the streams and rivers of DENA remain woefully understudied. A similar situation no doubt entails in WRST and YUCH, but there is no comparable literature available for comparison in those park units.

Table 5. List of new aquatic invertebrate taxa identified from 2011 CAKN samples. A * indicates a taxon collected in DENA that has not been previously reported (Conn 1998, Milner et al. 2003).

Order	Taxa
Plecoptera (stoneflies)	<i>Diura bicaudata</i> <i>Arcynopteryx compacta</i> *
Ephemeroptera (mayflies)	<i>Cinygma</i> sp <i>Baetis tricaudatus</i> * <i>Baetis foemina</i> * <i>Baetis phoebus</i> *
Trichoptera (caddisflies)	<i>Dicsomoecus obscuripennis</i> <i>Hydroptilidae</i> (a new family of caddisflies) <i>Ochrotrichia</i> sp. <i>Rhyacophila Alberta</i> Gr. (a species complex)
Diptera (true flies)	<i>Culicoides</i> sp. <i>Sublettea</i> sp. <i>Pedicia</i> sp.* <i>Cardiocladius albiplumis</i> * <i>Rheotanytarsus</i> sp.*
Mollusca (mollusks)	<i>Physidae</i> (a new family of snails)*

A total of 166 unique macroinvertebrate taxa have now been identified in the six 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. At least three of these taxa have never been formally identified, and would represent entirely 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 three sites in 2007, was collected again at four sites in 2008, four sites in 2009, and two sites in 2010. The midge has been collected in three years of sampling (2007–2009) from Hogan Creek (DENA), and in both 2008 and 2009 from 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. In contrast, in 2011 it was not collected from any sites in DENA (neither Little Stony Creek nor Hogan Creek was sampled), but was identified from two sites in WRST. The previously unidentified midge from the subfamily Tanytarsini that was collected from a stream in WRST in 2008 has not been found in any samples collected in 2009–2011. Lastly, a previously unidentified member of the family Empididae (dance flies) was collected at seven sites in 2009, and was collected again from several sites in 2010. 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. In 2011, this taxon was not identified in any of the samples collected. Nevertheless, in 2011 an attempt was made to field preserve a subset of specimens for DNA analysis. This effort will continue in 2012 and it should be possible to identify this taxon in the near future.

Benthic Diatoms

In 2011, 25 diatom samples were collected from 23 unique sites. As has been observed in previous years, species richness for diatoms was in general substantially higher than for macroinvertebrates, and varied from a low of four species (Stuver Creek tributary in WRST) to a high of 54 species (Hanagita River in WRST), with a mean and median of 28 species per site. A total of 460 diatom species have now been identified through the CAKN program; 166 of these were first identified in samples collected in 2006, 68 additional species were first identified in the 2007 samples, 126 in 2008, 36 in 2009, 45 in 2010, and another 20 in 2011. Given that much of the CAKN remains relatively unstudied, we expect to continue to add to this total for many years to come. In general, the lowest species richness and density are 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 in DENA has consistently supported 12–22 species of diatoms during the summer, as opposed to only three to five macroinvertebrate taxa. Diatom densities documented in 2011 varied from a low of 2.4×10^4 cells/cm² at the high-elevation tributary to Stuver Creek in WRST to a high of 4.2×10^7 cells/cm² at the small, stable meadow stream Paco Creek in WRST (located at Mile 27 of the Nabesna Road). The mean density across all sites sampled was 2.1×10^7 cells/cm².

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. In 2011, *Didymosphenia* was identified at 3 of the 23 sites sampled –Igloo Creek in DENA, Tanada Creek in WRST, and the upper Charley River in YUCH. In all three cases, the observed density was low, which is consistent with findings 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 (divided by Sable Pass). In 2010, *D. geminata* blooms were evident at both of these sites, but were patchy in nature and much less obvious than during previous blooms. No *D. geminata* blooms were observed in 2011. 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

CAKN has sampled a glacial river in both the summer and the fall since 2006, although no sampling was conducted in 2011. 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 2007–2010, the river was sampled in late August, when it was turbid, and exhibited very dynamic hydrology, and then in 2008–2010 sampled again in late September, when the water was clear and the flow was low and stable.

The results have been remarkable (Table 6.) It is not known exactly when the melt season ended for the East Fork Glacier in any of these years, 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 the glacial melt season. High total phosphorous accompanied by low soluble reactive phosphate is typical of the turbid glacial rivers that have been examined in CAKN. The likely explanation for this lack of interseasonal variability in water chemistry is a significant contribution to the shoulder season baseflow from groundwater sources with a substantial glacial component, although this has not been tested.

Table 6. Summary of instream conditions at East Fork Toklat River in summer vs. fall. Summer conditions represent a four-year average (2007–2010), and fall conditions are a three-year average (2008–2010).

Metric	Summer Value	Fall Value
Temperature (instantaneous)	6.7°C	3.1°C
Specific conductivity	424 $\mu\text{S}/\text{cm}$	462 $\mu\text{S}/\text{cm}$
Total nitrogen	145 $\mu\text{g}/\text{L}$	157 $\mu\text{g}/\text{L}$
Total phosphorous	108 $\mu\text{g}/\text{L}$	14 $\mu\text{g}/\text{L}$
pH	8.3	8.2
Nitrate-nitrogen	147 $\mu\text{g}/\text{L}$	155 $\mu\text{g}/\text{L}$
Phosphate-phosphorus	<1 $\mu\text{g}/\text{L}$	<1 $\mu\text{g}/\text{L}$
Macroinvertebrate taxon richness	6	10.3
Macroinvertebrate density	24 / m^2	73 / m^2
Diatom species richness	21	13.5
Diatom density	37,500/ cm^2	493,244/ cm^2

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 versus 2009 and 2010. In 2008, species richness was higher in the fall (15) than in the summer (12), whereas in 2009 and 2010 richness was higher in the summer (30 and 22, respectively) than in the fall (12 and 17, respectively). 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 300%, although it remains low relative to nonglacial streams. Presumably, macroinvertebrates spend the summer high-water season in hyporheic refugia and migrate to the surface bed when flows stabilize, although the nature of the signal that triggers such migration is unclear. 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.

Ongoing Evaluation of the Comparability Between LTEM and CAKN Macroinvertebrate Data

An important focus for the 2007 field season was to test whether legacy invertebrate data collected in DENA as part of the Long Term Ecological Monitoring Program (LTEM) would be compatible with data collected using the 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, that program collected macroinvertebrate data in 1994–1996 from a total of 58 streams located throughout DENA. These data could 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 the LTEM program versus those used by the CAKN program have the potential to complicate efforts to synthesize these two data streams. Initial comparative analyses based on data collected nearly simultaneously by both groups at 10 sites along the park road in 2007 revealed substantial differences in the results produced by the two programs. Both adjusted taxa richness and community composition differed significantly. Two of these sites have been resampled by the CAKN program in all four years from 2008 to 2011, two have been resampled three times, four have been resampled twice and all sites have been resampled at least once (Table 7). The original 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. Without this assumption, the comparability between the two datasets is reduced further. 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. The most substantial of these differences is that in the LTEM data, chironomid midges are left at the family level, whereas in the CAKN data midges are identified to genus. This difference results in 46 additional unique taxa in the CAKN data.

Table 7. LTEM sites that were resampled by CAKN in 2007–2011.

Stream Name (reach length in meters)	Description of CAKN Sampling Reach	Years Sampled
Hogan Creek (150 m)	Reach starts ~50 m above road	2007–2009
Sanctuary River (500 m)	Reach starts opposite pullout ¼ mile above bridge	2007, 2008, 2010
Highway Pass Creek (150 m)	Reach starts opposite gully/pullout	2007, 2010, 2011
Igloo Creek (250 m)	Reach starts ~100 m above bridge	2007–2011
Tattler Creek (150 m)	Reach starts ~50 m above road	2007, 2008, 2011
East Fork Toklat River *	Reach starts ~50 m above tributary confluence	2007–2010
East Fork Toklat tributary (200 m)	Reach starts ~200 m above confluence	2007–2010
Little Stony Creek (150 m)	Reach starts ~50 m above road	2007–2009
Moose Creek (500 m)	Top of reach at Kantishna bridge	2007–2011
Savage River (500 m)	Top of reach ~200 m downstream of bridge	2007, 2009

*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.

After making these adjustments, it was found that mean taxa richness was still substantially higher in the CAKN samples than in the LTEM samples (37% on average), while mean densities were similar. 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 habitat diversity of the larger stream segment. Furthermore, the LTEM program collects and composites 5 subsamples, whereas the CAKN program collects 8 subsamples. This effect should be more pronounced in larger streams, as was observed. The 2008–2011 CAKN data continue to support this idea in general. On average, the mean (2007–2011) observed richness in the CAKN samples is 63% higher than that observed in the 2007 LTEM samples (Table 8), which suggests that the 2007 results were not aberrant. The pattern of differences is less obvious than it was for the 2007 data however. Although the discrepancy is proportionally large for all larger streams (Sanctuary River, Igloo Creek, EF Toklat River, Moose Creek, Savage River), and is small in two relatively small streams (Hogan Creek, Little Stony Creek), large proportional differences are also evident at other smaller streams (Tattler Creek, EF Toklat tributary, Highway Pass Creek). Nevertheless, it remains likely that these differences are largely a result of the CAKN sampling methodology capturing a greater proportion of the habitat diversity at each site.

It is interesting to note that mean densities in the multiyear CAKN samples are substantially higher than in either the LTEM or CAKN samples collected in 2007. The 2007 CAKN samples were on average of similar density as the LTEM samples. Examination of the CAKN data shows that densities at almost all sites were much lower in the 2007 samples than in subsequent years. The

exceptions were Hogan Creek and Savage River. For example, the invertebrate density in the Sanctuary River in 2008 was similar to 2007 but in 2010 was 100% higher. The source of these differences is still under investigation.

Table 8. Comparison of observed adjusted taxa richness (S) and density between CAKN and LTEM samples at LTEM sites in DENA 2007–2011. LTEM values are for 2007, CAKN numbers are mean values derived from all sample years for a given site. Densities are given as the number of organisms per square meter.

Stream name	S (LTEM)	S (CAKN)	Density (LTEM)	Density (CAKN)
Hogan Creek	12	12.7	3548	6510
Sanctuary River	12	19.3	203	331
Igloo Creek	11	19.4	431	1581
Tattler Creek	10	17.3	517	1427
East Fork Toklat River*	3	5.0	19	24
East Fork Toklat tributary	8	18.5	263	1618
Little Stony Creek	10	12.6	456	5104
Moose Creek	18	27.8	1333	1988
Savage River	13	21.5	374	644
Highway Pass Creek	5	13.0	127	507
<i>Mean across all samples</i>	10.2	16.6	727	1970

*For CAKN samples, only values for summer samples are included in the comparison, since fall samples have much higher density and richness.

We also re-examined the species composition of the various samples. In 2007, the mean Sørensen 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). Sørensen 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. Hence a mean Sørensen dissimilarity of 0.34 may be interpreted to mean that on average, samples collected at the same sites several days apart by the CAKN and LTEM programs in 2007 shared 66% of their taxa in common. Although this mean between-sample dissimilarity (within-stream, between-program) was about half of the mean between-stream dissimilarity, it was still higher than expected. In 2006, the CAKN program resampled a single reach (Chalk Creek in WRST) five times in one day. The mean Sørensen 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 dissimilarity among these DENA streams. A similar exercise was conducted by the CAKN program 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 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 identified (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 2008–2011 are included in the analysis, it was found that the mean Sørensen dissimilarity between samples collected at these 8 streams in different years (0.36) was less than the mean distance between LTEM and CAKN samples collected at those streams only a few days apart in 2007 (0.40). Dissimilarities between the LTEM samples and both the 2008 (0.49) and 2009 (0.37) CAKN samples were similar to the 2007 results. These patterns have remained consistent through the 2011 CAKN data. Most of this difference was again driven by the substantial dissimilarity in samples collected in larger rivers. 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. Accordingly, for the time being, invertebrate data collected by the LTEM program have not been included in any analyses of CAKN invertebrate data.

Initial Analyses of Year-to-Year Variability

Water Chemistry

2011 was the sixth year of data collection (fifth for most sites), which provides enough data to examine year-to-year variability in some of the metrics collected (Table 9). For the sonde data (temperature, specific conductivity, pH and dissolved oxygen), there are 35 sites with data from at least two years, 25 sites with data from at least three years, 19 sites with data from at least four years, 11 sites with data from at least five years, and 3 sites with data from all six years. For laboratory-derived water chemistry data, there are 37 sites with data from at least two years, 18 sites with data from at least three years, 11 sites with data from at least four years, 6 sites with data from at least five years, and 3 sites with data from all six years. In addition, a number of sites have data recorded multiple times per year. Although power to detect trends depends on various factors, an 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 that there is a wide range in the variability of water chemistry (as expressed by the coefficient of variation), both across sites and among different chemical parameters. pH values are fairly constant across years at all sites, but variance in pH cannot be easily evaluated, so pH is not included. In general specific conductance shows the least interannual variability (mean across all streams = 0.23), and total phosphorous shows the greatest (mean = 0.78). However, the variation among streams for a given parameter can be substantial. For example, the minimum observed CV in total nitrate nitrogen is 0.014, which is quite low, whereas the maximum value is 1.1,

nearly 100-fold greater. For most streams used to calculate the statistics in Table 9 only three years of data are available, so caution must be used in drawing conclusions.

Table 9. 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.

Parameter	Minimum CV	Maximum CV	Mean CV
Specific conductance	.02	.43	.23
Total alkalinity	.13	.77	.39
Total nitrogen	.028	1.4	.43
Nitrate-nitrogen	.014	1.1	.49
Total phosphorous	.18	2.5	.78
Sulfate-sulfur	.27	1.3	.63
Dissolved organic carbon	.05	1.4	.54

Three sites have water chemistry data available for all six years of the monitoring program. Table 10 shows the mean, 95% confidence interval (95% CI) and coefficient of variation (CV) for some key water chemistry parameters at these sites (Rock Creek, Chalk Creek and Jack Creek in WRST). All three sites were also visited more than once per year in at least one of the six years. Chalk Creek was visited twice in 2006 and 2007, four times in 2008, five times in 2009, four times in 2010, and five times in 2011; Rock Creek was visited twice in 2006, once in 2007, four times in 2008, three times in 2009, three times in 2010, and five times in 2011; and Jack Creek was visited once in 2006, once in 2007, three times in 2008, four times in 2009, four times in 2010, and five times in 2011. 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.

Table 10. Visit-to-visit variability in water chemistry parameters at sites sampled at least once per year from 2006 to 2011.

Stream	Parameter	N	Mean	95% CI	CV
Rock Creek	Specific conductance	17	355 $\mu\text{S}/\text{cm}$	+/- 38	0.22
Chalk Creek	Specific conductance	20	443 $\mu\text{S}/\text{cm}$	+/- 22	0.11
Jack Creek	Specific conductance	18	389 $\mu\text{S}/\text{cm}$	+/- 23	0.13
Rock Creek	Total alkalinity	11	150 mg/L	+/- 9	0.11
Chalk Creek	Total alkalinity	13	157 mg/L	+/- 10	0.11
Jack Creek	Total alkalinity	12	126 mg/L	+/- 13	0.18
Rock Creek	Total nitrogen	11	185 $\mu\text{g}/\text{L}$	+/- 31	0.28
Chalk Creek	Total nitrogen	13	251 $\mu\text{g}/\text{L}$	+/- 27	0.20
Jack Creek	Total nitrogen	12	215 $\mu\text{g}/\text{L}$	+/- 17	0.14
Rock Creek	Dissolved organic carbon	11	3.61 mg/L	+/- 0.51	0.24
Chalk Creek	Dissolved organic carbon	12	1.64 mg/L	+/- 0.49	0.53
Jack Creek	Dissolved organic carbon	12	1.96 mg/L	+/- 0.40	0.36
Rock Creek	Total phosphorous	11	8 $\mu\text{g}/\text{L}$	+/- 4	0.78
Chalk Creek	Total phosphorous	13	16 $\mu\text{g}/\text{L}$	+/- 4	0.41
Jack Creek	Total phosphorous	12	15 $\mu\text{g}/\text{L}$	+/- 7	0.76
Rock Creek	Nitrate-nitrogen	12	58 $\mu\text{g}/\text{L}$	+/- 14	0.44
Chalk Creek	Nitrate-nitrogen	12	163 $\mu\text{g}/\text{L}$	+/- 24	0.26
Jack Creek	Nitrate-nitrogen	12	127 $\mu\text{g}/\text{L}$	+/- 18	0.25
Rock Creek	Sulfate-sulfur	11	17.0 mg/L	+/- 2.2	0.22
Chalk Creek	Sulfate-sulfur	12	24.7 mg/L	+/- 1.9	0.14
Jack Creek	Sulfate-sulfur	12	24.2 mg/L	+/- 1.1	0.08

Note that for Rock Creek and Chalk Creek, statistics except for pH were calculated without including measurements made during a major flood event in 2007.

Alkalinity and pH show relatively low variability at all three sites, whereas dissolved organic carbon and total phosphorous are quite variable at all three sites. Much of the variability observed in water chemistry is due to the effects of changing discharge on the importance of different flow paths and source waters. During high flow, for example, overland and shallow subsurface flows become more important, while at baseflow the relative contribution of groundwater may be increased. Accounting for changes in discharge can help to reduce the observed variability in some water chemistry parameters. The CAKN Stream Monitoring Program does not routinely quantify discharge during site visits (although qualitative assessments of flow are recorded); however, it can be partially accounted for by using season as a surrogate. In spring and early summer, discharge is generally

much higher than later in the summer, when baseflow is more commonly observed. The effect of considering spring measurements separately from summer measurements is shown in Table 11 for specific conductivity. In nearly every case, the visit-to-visit variability is reduced substantially for season-specific samples compared to all samples. This suggests that by accounting for variations in discharge either by restricting analyses to same-season sampling, or by using the qualitative flow descriptions we routinely record to “bin” measurements, we will greatly increase our power to detect trends in water chemistry. Similar results were found for other water chemistry parameters (data not shown).

Table 11. Effects of seasonality on variability (as represented by the coefficient of variation – CV) of specific conductivity measurements for selected streams. ND indicates that insufficient data exist to calculate a CV.

Stream	CV (all samples)	CV (spring only)	CV (summer only)
Rock Creek DENA	.224	.005	.009
Little Stony Creek	.410	.152	.090
Tattler Creek	.361	.103	.087
Hogan Creek	.199	.108	.106
Moose Creek	.071	.034	.055
EF Toklat tributary	.338	.134	.086
EF Toklat River	.129	.224	.049
Stony Creek	.547	.220	.129
Igloo Creek	.320	.033	.072
Sanctuary River	.160	.144	.082
Chalk Creek	.156	.099	.147*
Rock Creek WRST	.217	.027	.071*
Little Jack Creek	.207	ND	.170
Jack Creek	.140	.100	.148
Mean	.247	.070	.098

*A major flood event in July 2007 was ignored for this calculation.

Macroinvertebrates

A preliminary analysis of patterns in macroinvertebrate density and community composition over the six-year sampling period was also conducted. Briefly, abundance data were converted to presence/absence to reduce the effects of extreme variations in abundance, and a Bray-Curtis dissimilarity matrix was calculated. Bray-Curtis dissimilarity is identical to Sørensen dissimilarity when applied to presence/absence data. Both indices vary between 0 and 1, with a value of 0 indicating identical taxa compositions, and a value of 1 indicating that the two samples share no taxa in common. An important caveat is that calculations of dissimilarity can be affected by differences in

sample size, which is relevant here because sample sizes were not standardized beforehand. This decision was made based on the large variation in abundances observed in the dataset, which would have required either removing a number of sites, or in standardizing to a very low abundance. Nevertheless, this analysis is useful for preliminary data exploration. The matrix itself was examined to assess year-to-year variation in composition at sites with multiple years of data (Table 12); in addition, cluster analysis and non-metric multidimensional scaling (NMDS) ordination were used to look at overall patterns among sites and across years.

Table 12. Year-to-year variation in macroinvertebrate community composition, based on taxon presence/absence with rare species excluded. Figures given are Sørensen dissimilarities and are averages among all year pairs (e.g., 2006–2007, 2006–2008, 2006–2009, 2006–2010, 2007–2008, 2007–2009, 2007–2010, 2008–2009, 2008–2010, 2009–2010 , etc. if data were available for all six years). The last column represents the mean within-year dissimilarity between summer and fall samples for sites sampled in both seasons.

Stream Name (N)	Summer/Summer	Fall/Fall	All Samples	Summer/Fall (within year)
Chalk Creek (12)	.27	.27	.29	.30
Rock Creek WRST (10)	.31	.34	.36	.37
EF Toklat River (7)	.72*	.53*	.59*	.56*
Jack Creek (7)	.25	.25	.29	.31
Moose Creek (5)	.21	--	.21	--
Igloo Creek (5)	.32	--	.32	--
Little Jack Creek (5)	.29	--	.38	--
Little Stony Creek (4)	.35	--	.32	.24
EF Toklat tributary (4)	.40	--	.40	--
Rock Creek DENA (3)	.30	--	.30	--
McKinley Bar Trail creek (3)	.20	--	.20	--
Willow Creek (3)	.40	--	.42	.47
Hogan Creek (3)	.24	--	.24	--
Sanctuary River (3)	.22	--	.22	--
Highway Pass Creek (3)	.40	--	.40	--
Tattler Creek (3)	.25	--	.25	--
Rufus Creek (2)	.22	--	.22	--
Savage River (2)	.28	--	.28	--
Beaver Creek (2)	.24	--	.24	--
Lake Creek WRST (2)	.37	--	.37	--
Gilahina River (2)	.26	--	.26	--

Table 12. Year-to-year variation in macroinvertebrate community composition, based on taxon presence/absence with rare species excluded. Figures given are Sørensen dissimilarities and are averages among all year pairs (e.g., 2006–2007, 2006–2008, 2006–2009, 2006–2010, 2007–2008, 2007–2009, 2007–2010, 2008–2009, 2008–2010, 2009–2010, etc. if data were available for all six years). The last column represents the mean within-year dissimilarity between summer and fall samples for sites sampled in both seasons (continued).

Stream Name (N)	Summer/Summer	Fall/Fall	All Samples	Summer/Fall (within year)
Gorge Creek spring (2)	.33	--	.33	--
McKinley Bar spring (2)	.19	--	.19	--
Upper Igloo Creek (2)	.27	--	.27	--
Lake Creek DENA (2)	.41	--	.41	--
Stony Creek tributary (2)	.33	--	.33	--
Upper Moose Creek (2)	.36	--	.36	--
Young Creek (2)	.47	--	.47	--
Travertine Stream (2)	.33	--	.33	--

*Due to the very low species richness at this site, the loss or gain of a single species can have a large effect on dissimilarity.

The site with the most invertebrate samples is Chalk Creek, where samples have been collected in all six 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 Sørensen 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%. However, in 2009, samples collected from Chalk Creek 17 days apart were substantially more similar than this (Sørensen dissimilarity = 0.095), so this may be a conservative estimate. Mean dissimilarity among all Chalk Creek samples was 0.29; 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. Mean dissimilarity between summer-only samples was 0.27, as was the mean dissimilarity among fall-only samples. Given the inferred sampling error of about 0.20, this suggests that variation in actual community composition may actually be rather low from year to year. Similar results were obtained at Little Stony Creek. Although it is clearly risky to draw broad conclusions based on only six years of data, 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 29 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 0.30. Mean year-to-year dissimilarity among fall samples was similar at 0.29. There was a substantial degree of variation among streams, however (Table 12), with interannual dissimilarities ranging from a low of 0.19 (McKinley Bar spring in DENA) to a high of 0.47 (Young Creek in WRST). Although only a limited number of fall samples have been collected, it appears that seasonal variation (mean within-year seasonal dissimilarity = 0.34) is similar to interannual variation among samples collected in the same season. 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.

Cluster analysis (Figure 5) provides another and potentially 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, 143 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. Rare taxa (those occurring at fewer than 5% of sites) were excluded from the analysis, as they tend to add noise and obscure relationships between samples. A very large percentage of OTU taxa in CAKN streams are rare according to this definition (71 of 137). This is reflective of the high beta-diversity among CAKN stream communities. Of the 137 unique taxa in the data set, only 11 have been collected in more than 50% of samples, and only 48 have been collected in more than 10% of samples. Over 20% of taxa occur in less than 1% of the samples. While this unusually high number of rare taxa might be expected to interfere with a robust classification, a cluster analysis that retained all taxa found at more than 1% of sites was broadly similar to the analysis depicted in Figure 5. Typically, ubiquitous taxa (those occurring at more than 95% of sites) are also excluded as they tend to obscure differences among samples. However, no taxa are found at more than 82% of sites in this data set. An important caveat to this analysis is that sample sizes were not standardized, which, as noted above, can potentially have a large effect on dissimilarity calculations, and hence on any classification procedure that is based on a dissimilarity matrix.

Inspection of the cluster dendrogram depicted in Figure 5 shows that in nearly all cases, samples from a given site collected in different years cluster together, even if the dendrogram is trimmed with most of the information 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. In a few 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. However, the general pattern appears to be that species composition remains relatively stable both within and across years at any given site, which is expected given what was found from examination of the distance matrix.

The delineation of groups or classes based on the results of a cluster analysis is somewhat arbitrary in that there are no explicit rules for group selection. However, one rule of thumb is that group cutoffs made with less information remaining (farther to the right or “higher up” on the dendrogram) are

more robust. Generally, cutoffs should be made with no more than about 50% information remaining. In the present case, nine groups were delineated (Figure 5). 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 tend to 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 three 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, which is also a lake outlet stream. In contrast, Lake Creek is only distantly related to Moose Creek, into which it flows and which is less than a mile away. The general lack of strictly geographic patterns in stream invertebrate communities, at least at moderate geographic scales, is consistent with the idea that local-scale environmental conditions play a major role in structuring lotic communities. One exception to this rule is that samples collected in streams that flow off of Sable Pass in DENA (Igloo Creek, Tattler Creek, and the EF Toklat River tributary) clustered tightly together (Group 2). The reason for this exception is not immediately apparent and warrants further investigation.

The uppermost 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 below this split (labeled “Glacial/unstable” in Figure 5) are either from glacial rivers or streams with intermittent flow or high-gradient streams with unstable beds. Other significant divisions are illustrated in Figure 5, and many seem to be related to ecosystem 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. At a slightly higher level on the dendrogram, these lake outlet streams also cluster with most of the spring-fed streams and other streams that appear hydrologically stable based on the prevalence of bryophytes. 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 in DENA.

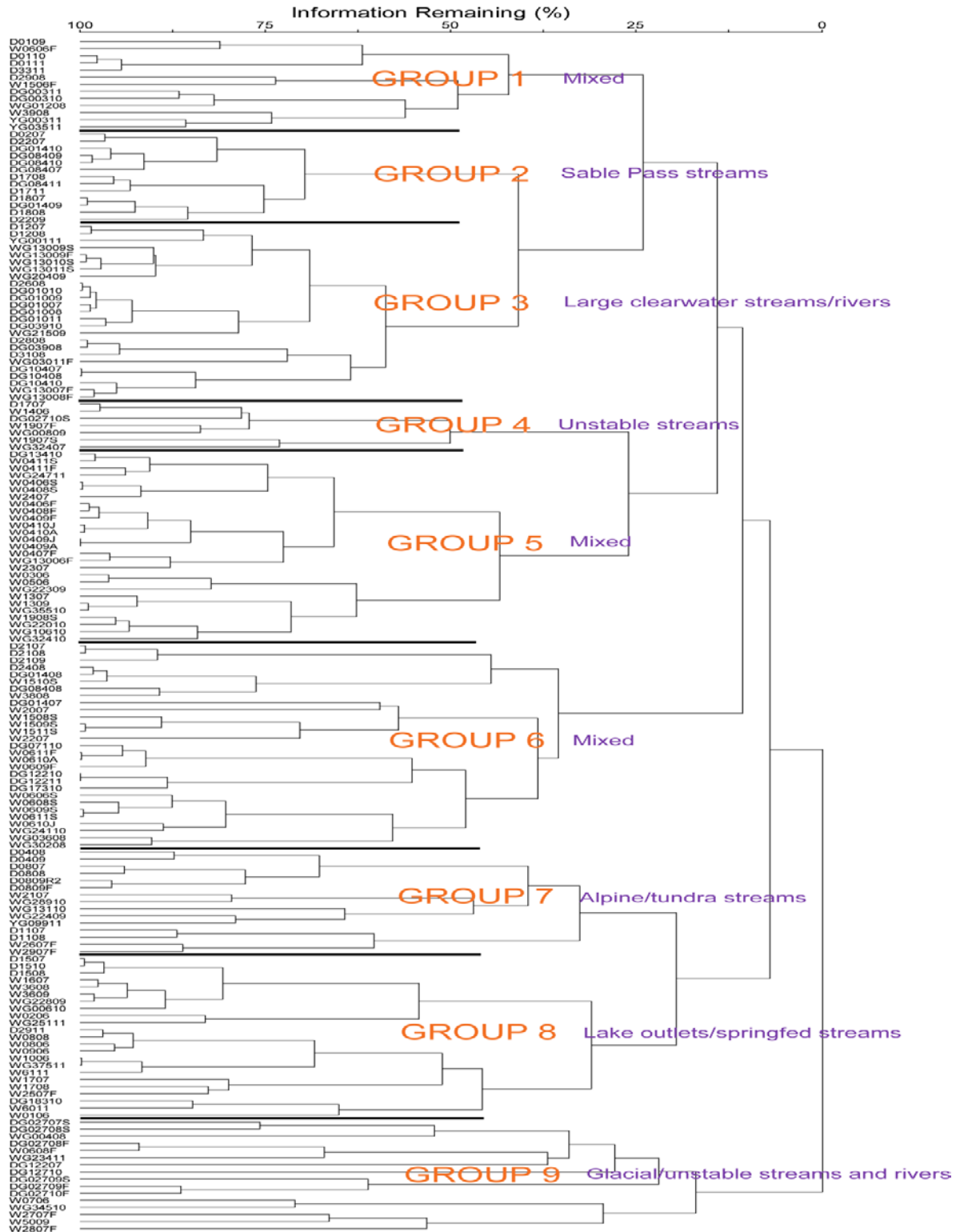


Figure 5. UPGMA cluster analysis of CAKN invertebrate data collected from 2006 to 2011. UPGMA with $\beta = -0.25$ was conducted in PC-Ord 6.0 using a Sørensen distance metric. Sample codes are abbreviated as follows: DENA01 sampled in 2009 is listed as D0109, etc. Details for sample codes are in Appendix A.

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 UPGMA, 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 Sørensen dissimilarity (Peck 2010). The dendrogram resulting from group-average clustering was similar in structure to the dendrogram depicted in Figure 5 (not shown). Similarly, the outcome can be affected by the dissimilarity measure selected to create the distance matrix on which the cluster analysis is based. To test for this, the clustering analysis was also conducted using both Jaccard dissimilarity and Euclidean distance as the distance measures, with very similar results (data not shown). These findings suggest that the classification is reasonably robust.

There are also a number of quantitative methods to test the strength of the group classification that is derived from a cluster analysis, including Classification Strength (Van Sickle and Hughes 2000), Analysis of Similarity (ANOSIM; Clarke 1993), and Multi-response Permutation Procedure (MRPP; Mielke and Berry 2001). All of these methods essentially compare the similarity of samples within groups to the similarity of samples across groups to test the null hypothesis that there is no difference between groups. MRPP is a permutation-based randomization procedure, and is perhaps the most widely used approach for ecological analyses. MRPP is non-parametric and therefore makes no distributional assumptions regarding the underlying community data. An MRPP analysis was conducted on the 9-group classification shown in Figure 5 using 10,000 permutations. The results showed that the classification was highly significant ($p < 1 \times 10^{-8}$). MRPP also generates a statistic called the A value (defined as the chance-corrected within-group agreement), which describes within-group homogeneity compared to random expectation. The A value can be thought of as a measure of effect size. For the 9-group classification of the CAKN data depicted in Figure 5, $A = 0.22$, which indicates a strong degree of within-group homogeneity (values for community data are typically < 0.1 and are rarely > 0.3). For datasets with large numbers of samples, the A value can be more informative than the p value, since a statistically significant p value can be obtained even when no ecologically-important effect is present. For example, an MRPP conducted on the same dataset but with samples grouped by park unit also yields a highly statistically significant classification ($p = 0.0000004$), but the A value in this case is only 0.01, indicating that the ecological significance of this classification is minimal.

Ordination is another set of methods commonly used to visualize complex community data. In particular, 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). NMDS essentially reduces the dimensionality of a complex high-dimension dataset to a small number of dimensions while maintaining the rank order of distances between the samples. An NMDS ordination of the CAKN invertebrate data from 2006 to 2011 is shown in Figure 6. As was the case for cluster analysis, the abundance data were converted to presence/absence, and rare species (those occurring in fewer than 5% of the samples) were deleted from the dataset prior to analysis. The best solution was three-dimensional (stress, a measure of how close to monotonic the relationship between the initial and final configuration is, was 18 and was significantly less than a solution with randomized data); however, only the first two dimensions are depicted in Figure 5. The first three

ordination axes of the ordination have a cumulative r^2 of 0.796, which indicates that much of the variation in the original data matrix is captured accurately.

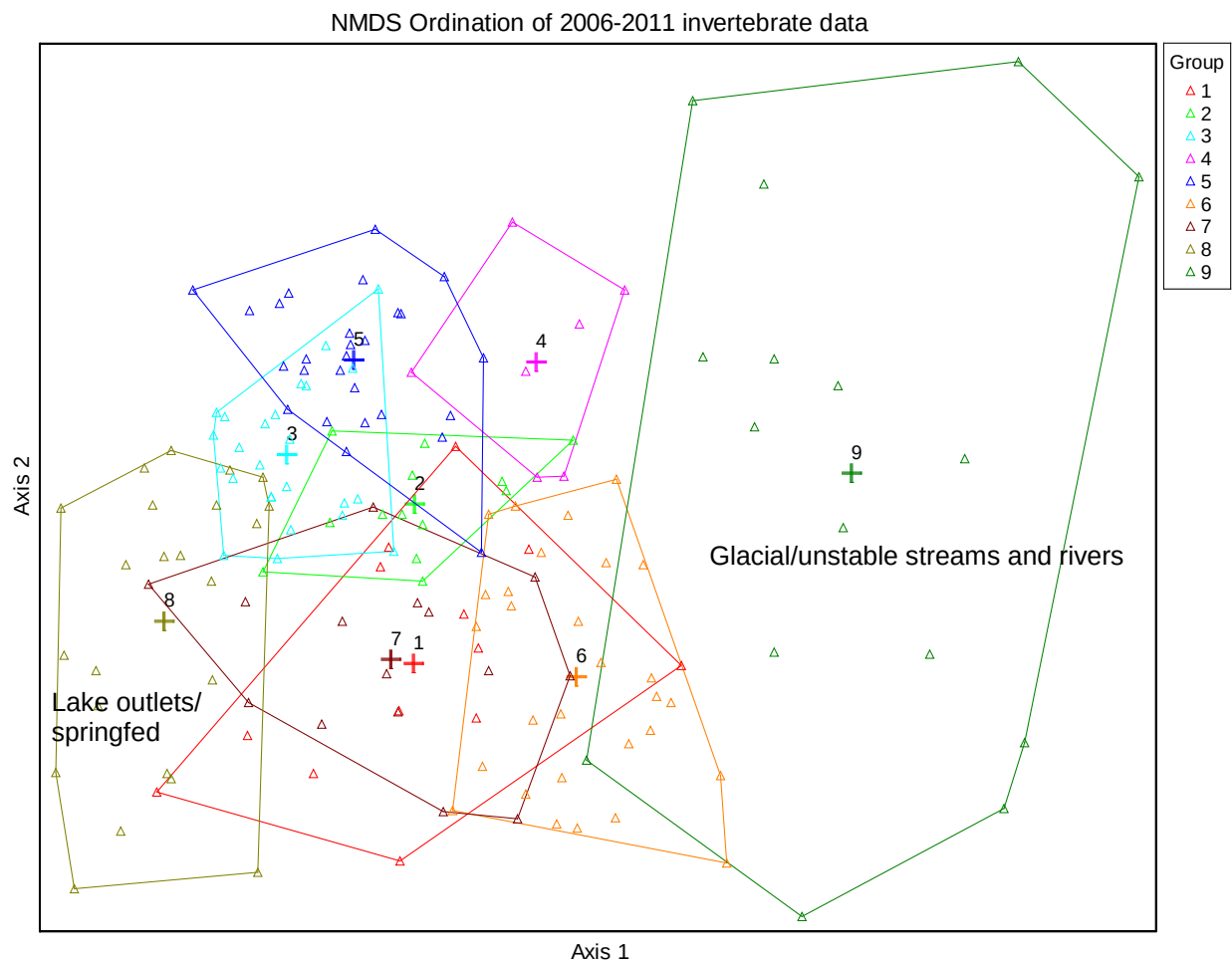


Figure 6. NMDS ordination of invertebrate data from 2006 to 2011. Colors and numbers correspond to the groups identified in the UPGMA cluster analysis. The + symbols indicate the locations of the centroids of each group in ordination space, the Group number is given next to each centroid. Convex hulls enclosing all points in each group are also depicted. NMDS was conducted in PC-Ord 6.0. A preliminary run was used to select the optimal number of dimensions (3) based on incremental stress reduction. A final run was conducted using 250 iterations, each starting from a random seed, to identify the optimal solution.

As was the case for cluster analysis, little or no geographic pattern is apparent, again with the exception of Group 2, which contains only samples from streams that drain Sable Pass in DENA. This suggests that differences in community composition among these sites are largely driven by local-scale environmental filters (*sensu* Poff 1997) rather than broad-scale biogeographic differences among regions. Ecologically similar sites tend to be grouped in the ordination, as in the cluster dendrogram. Two of these groups are labeled in Figure 6. Group 8 is composed of lake outlets and springfed streams with relatively stable flow regimes and temperature regimes (data not shown). As discussed above, these streams tend to have significant bryophyte and algae cover, and support high

densities of invertebrates. Group 9 is composed of samples from glacial rivers, intermittent streams, and high-gradient colluvial streams. These systems are characterized by highly variable flow regimes and/or unstable substrates and generally have sparse algal cover, no bryophytes and very low densities of invertebrates. These groups are located on opposite ends of Axis 1 (the horizontal axis in Figure 6), suggesting that this axis is correlated with some measure of ecosystem stability. The positions of the other groups in ordination space are also consistent with this idea. For example, Group 3 (large clearwater streams and rivers) is also composed of relatively stable systems, and is located near Group 8 in ordination space. Note that although all of the samples from glacial and unstable streams are found together at the right end of the horizontal axis, the ordination positions of these samples are distributed along the entire length of the vertical axis. 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. That Axis 1 explains the preponderance of variation in the ordination ($r^2 = 0.515$) suggests that ecosystem stability, as expressed through a combination of flow regime, temperature regime and substrate stability, is a major determinant of community composition among CAKN streams, results that are again broadly consistent with those from Milner's long-term study of streams along the Denali Park Road (Milner et al. 2006).

In addition to providing insight into ecological patterns among sites (spatial variation), NMDS can be used to evaluate temporal variation and trends in community composition at individual sites. In Figure 7, all of the samples collected between 2006 and 2011 at the two sites with the most data (Rock Creek and Chalk Creek in WRST) are circled. It can be seen that the variation between samples collected at these sites at different times (temporal variation) is substantially less than the observed variation between all sites (spatial variation), and even between sites within the same biologically-based group, which consistent with what other analyses have indicated. To test for potential directional shifts in species composition over time in CAKN streams, successional vector analysis was conducted for all sites (14 in total) with at least three years of invertebrate data. This method can reveal whether year-to-year changes in composition are directional or locally random in ordination space. Directional changes in ordination space may indicate an ongoing shift in composition, i.e., a trend, whereas more random patterns suggest interannual fluctuations in species composition without directional change. None of the 14 sites analyzed showed any directional change, based on successional vectors. The successional vectors for Rock Creek and Chalk Creek are also shown in Figure 7 as blue arrows connecting samples from successive years. As more data are collected, these methods will become more useful in assessing both spatial and temporal patterns in the distribution of invertebrates.

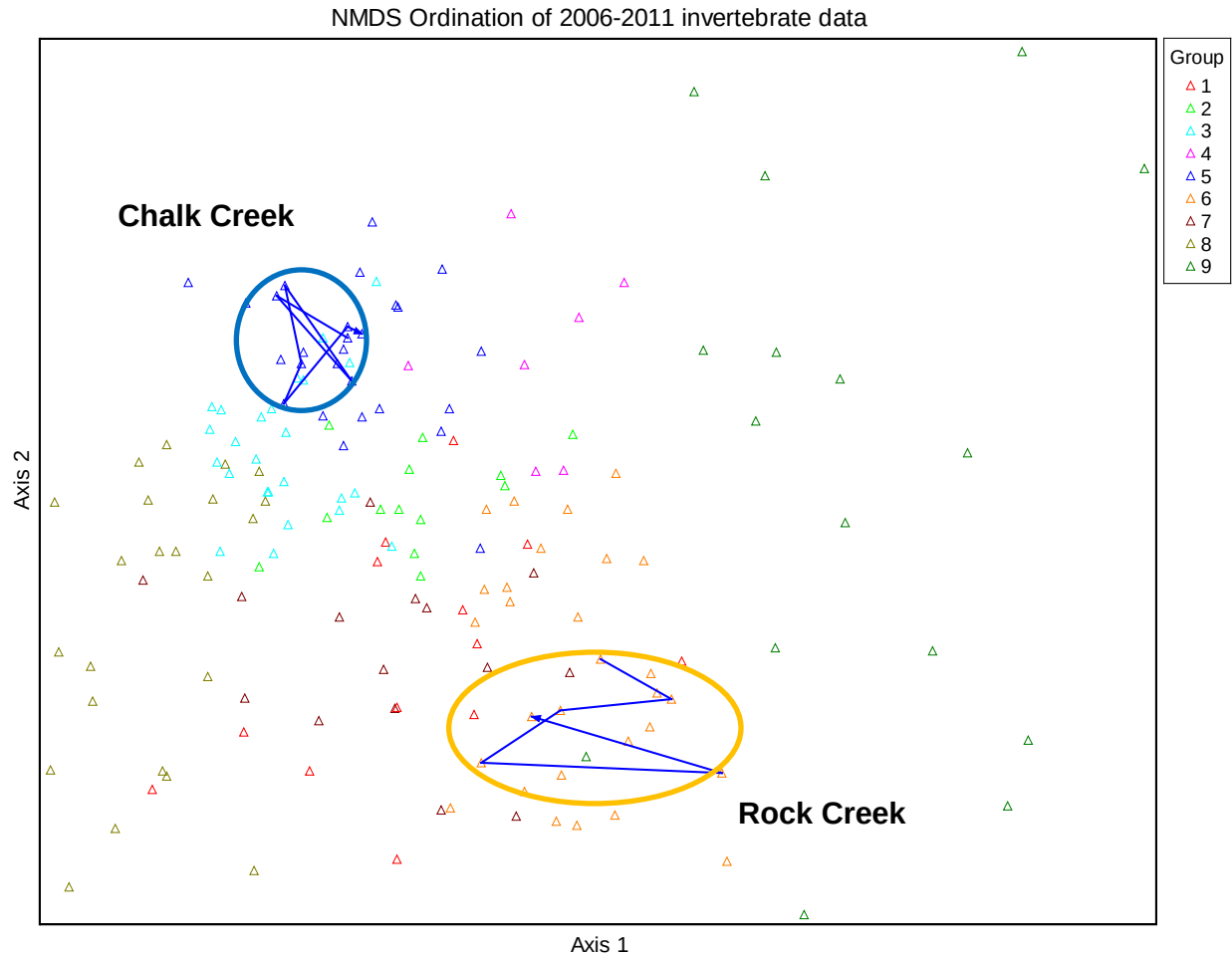


Figure 7. NMDS ordination of invertebrate data from 2006 to 2011 showing successional vectors for Rock Creek and Chalk Creek. Methods are the same as described in the legend to Figure 6.

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” modeling, after the original RIVPACS model developed in Great Britain in the 1980s (see e.g., Clarke et al. 2002), although more recently it is also referred to as “O/E” modeling. This method is part of a family of assessment techniques known more broadly as the Reference Condition Approach (Bailey et al. 2004), in which the biological community at a site to be assessed is compared to the community at an appropriate set of reference sites (sites that are relatively unimpaired by human activity). 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 biologically similar 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 began in 2011 to explore the utility of these methods for detecting change in network stream ecosystems. A preliminary model, constructed using data collected from 2006 to 2009, was presented in the CAKN 2009 Annual Report (Simmons 2011). The following discussion concerns our attempt to improve on the performance of the initial model, using data collected through 2011. An important change is that data from glacial and intermittent streams, which were outliers in the previous modeling exercise due

to their depauperate invertebrate fauna, were excluded in this analysis. Using macroinvertebrate data (166 taxa) from 70 calibration sites in DENA and WRST, we have recently “upgraded” the 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).

Cluster analysis (flexible- β UPGMA) was used to separate the 70 sites into 6 groups based on biological similarity, as described above. The random forests method (Breiman 2001, Cutler et al. 2007) was used to develop a discriminant model that could predict new sites into the biological groups based on environmental gradients, and the discriminant model was used to calculate posterior probabilities of group membership for each calibration site. Ideally this procedure would be repeated with a separate 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 13 lists the globally important predictor variables from the random forest discriminant analysis.

Table 13. List of globally important predictor variables from random forest discriminant analysis of CAKN macroinvertebrate data. Climate variables are derived from PRISM climate models (Daly et al. 2008). The values for all variables are calculated for the entire drainage basin rather than the sampling reach. Listing is in order of importance value.

Variable Name	Interpretation
Watershed Area	Total watershed area above sampling location
Watershed Perimeter	Total watershed perimeter above sampling location
Annual Max Temp	Average annual maximum air temperature in watershed (from PRISM)
July Max Temp	Average July maximum air temperature in watershed (from PRISM)
Annual Min Temp	Average annual minimum air temperature in watershed (from PRISM)
CV Slope	Coefficient of variation in watershed slope
Min. Elevation	Minimum watershed elevation (generally same as sampling location)
Percent Evergreen	Percent of landcover in watershed classified as evergreen forest
Percent Lakes	Percent of landcover in watershed classified as lake
Percent Wetland	Percent of landcover in watershed classified as wetland

There are three broad categories of environmental descriptors represented — stream power (watershed size and slope), climate, and land cover, all of which are important determinants of stream ecosystems. They are also similar to the variables that were selected in the pilot modeling effort. 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 distinguishing among 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. A series of R scripts developed by USEPA was used to calculate the probabilities of capture for each taxon at each reference site that was evaluated, to calculate E, and to calculate O/E (available at <http://www.epa.gov/wed/pages/models/rivpacs/rivpacs.htm>).

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 positive bias to the model in that on average it predicts fewer taxa than were actually observed at individual sites. This slight positive bias has been observed by other researchers when using random forest discriminant models (C.P. Hawkins, personal communication). The precision of the model, as represented by the standard deviation of reference site O/E scores, was 0.17, which is a substantial improvement over the initial model (SD = 0.24) and is comparable to other large-scale RIVPACS models constructed using data from the conterminous United States and in other countries (J.D. Ostermiller, personal communication). One factor affecting the precision of the model is the combination of very high heterogeneity and a relatively small number of reference sites. Taxa with a $P_c < 0.5$ (that is, relatively rare taxa) are normally ignored for calculation of E as a way of improving model performance (Hawkins et al. 2000, Ostermiller and Hawkins 2004). This approach is somewhat problematic for the CAKN data, because most of the taxa in the dataset are found at only relatively few sites, and 2/3 of the taxa never had a probability of capture (P_c) above 0.5 at any site. This makes generating accurate predictions challenging. Increasing the number of sites in the reference data set going forward should alleviate this effect to some degree.

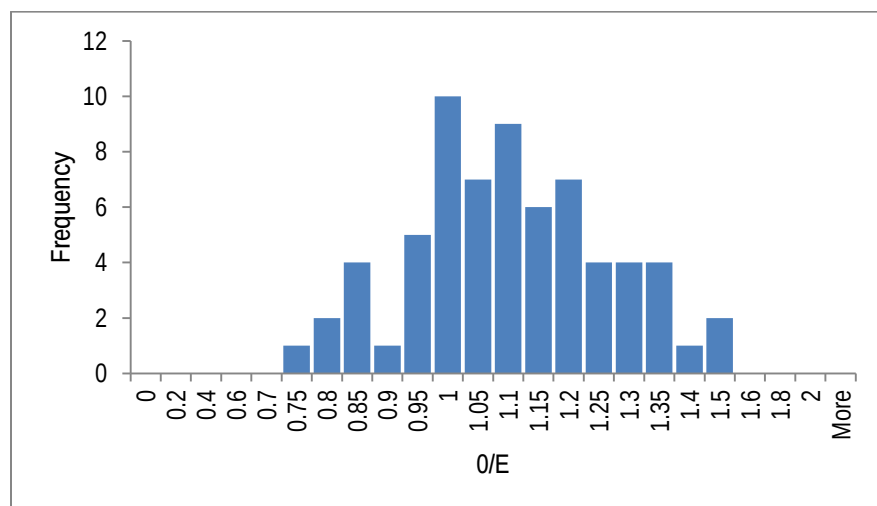


Figure 8. Distribution of O/E scores for calibration sites. Mean = 1.07, SD = 0.17.

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 70 calibration sites. For an unbiased model, O should be equally distributed above and below E for any predicted value of E. That is a plot of O versus E should be equivalent to a 1:1 line with a high r^2 . 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 again indicates a slight positive model bias, particularly for more diverse streams. In other words, at sites with high taxa richness, the model is consistently underpredicting the number of taxa (i.e., the number of expected taxa that are actually observed at these sites is systematically higher than the number predicted). 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.

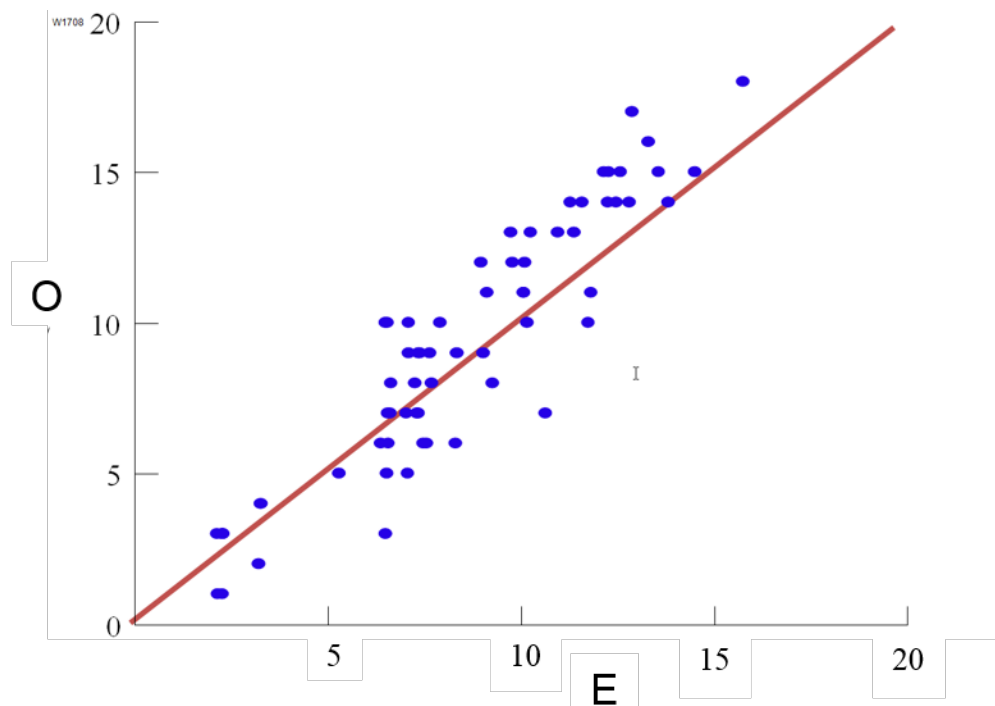


Figure 9. Plot of O vs. E for calibration sites. $R^2 = 0.85$, not significantly different from the 1:1 line.

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.

The results have also made clear that more data are necessary to improve the precision and sensitivity of the models, and to reduce bias. Therefore, to increase the number of sites available, we have obtained additional data from both USEPA (46 sites) and the U.S. Geological Survey (21 sites). Although many of these sites are not within the boundaries of the CAKN parks, they are generally in the geographic gap (the Tanana Basin) that separates all three park units in the network. Inclusion of these samples should substantially improve the geographic continuity of data coverage in addition to densifying the range of ecological conditions represented in the data set. We are currently working on a 4th iteration of the model incorporating these additional data. We will also be testing the sensitivity of the model to ecological impairment by utilizing data collected from sites that are presumed to be altered by anthropogenic activity. These include several sites in the Kantishna Hills of DENA that were affected by historical dredge and hard rock mining; Coal Creek in YUCH, which was heavily altered by decades of dredge mining; as well as a number of sites downstream from road crossings and other park infrastructure. This is the first step towards applying the model to the comprehensive evaluation of streams in all three park units that may be altered due to the effects of historical or present-day human activity.

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Appendix A

Abbreviated sitecodes used in cluster dendrogram (Figure 5) and ordination (Figure 7)

Abbreviation	Original Sitecode	Site Name	Park	Year	Season
D0109	DENA-001	Rock DENA	DENA	2009	Summer
D0110	DENA-001	Rock DENA	DENA	2010	Summer
D0111	DENA-001	Rock DENA	DENA	2011	Summer
D0207	DENA-002	E. Savage River spring	DENA	2007	Summer
D0408	DENA-004	Travertine stream	DENA	2008	Summer
D0409	DENA-004	Travertine stream	DENA	2009	Summer
D0807	DENA-008	Little Stony Creek	DENA	2007	Summer
D0808	DENA-008	Little Stony Creek	DENA	2008	Summer
D0809R2	DENA-008	Little Stony Creek	DENA	2009	Summer
D0809F	DENA-008	Little Stony Creek	DENA	2009	Fall
D1107	DENA-011	Gorge Creek spring	DENA	2007	Summer
D1108	DENA-011	Gorge Creek spring	DENA	2008	Summer
D1207	DENA-012	McKinley Bar spring	DENA	2007	Summer
D1208	DENA-012	McKinley Bar spring	DENA	2008	Summer
D1507	DENA-015	McKinley Bar Tr. creek	DENA	2007	Summer
D1508	DENA-015	McKinley Bar Tr. creek	DENA	2008	Summer
D1510	DENA-015	McKinley Bar Tr. creek	DENA	2010	Summer
D1707	DENA-017	Tattler Creek	DENA	2007	Summer
D1708	DENA-017	Tattler Creek	DENA	2008	Summer
D1711	DENA-017	Tattler Creek	DENA	2011	Summer
D1807	DENA-018	Igloo Cr abv Tattler	DENA	2007	Summer
D1808	DENA-018	Igloo Cr abv Tattler	DENA	2008	Summer
D2107	DENA-021	Hogan Creek	DENA	2007	Summer
D2108	DENA-021	Hogan Creek	DENA	2008	Summer
D2109	DENA-021	Hogan Creek	DENA	2009	Summer
D2207	DENA-022	Savage River	DENA	2007	Summer
D2209	DENA-022	Savage River	DENA	2009	Summer
D2408	DENA-024	Gorge Creek	DENA	2008	Summer
D2608	DENA-026	Moose Creek @ crossing	DENA	2008	Summer
D2808	DENA-028	Moose Creek airstrip	DENA	2008	Summer
D2908	DENA-029	Lake Creek DENA	DENA	2008	Summer
D2911	DENA-029	Lake Creek DENA	DENA	2011	Summer
D3108	DENA-031	Cantwell Creek	DENA	2008	Summer
D3311	DENA-033	Hines Creek trib	DENA	2011	Summer
DG00311	DENA-GRST-003	Stony Creek trib	DENA	2011	Summer
DG00310	DENA-GRST-003	Stony Creek trib	DENA	2010	Summer
DG01007	DENA-GRTS-010	Moose Creek	DENA	2007	Summer
DG01008	DENA-GRTS-010	Moose Creek	DENA	2008	Summer

Abbreviation	Original Sitecode	Site Name	Park	Year	Season
DG01009	DENA-GRTS-010	Moose Creek	DENA	2009	Summer
DG01010	DENA-GRTS-010	Moose Creek	DENA	2010	Summer
DG01011	DENA-GRTS-010	Moose Creek	DENA	2011	Summer
DG01407	DENA-GRTS-014	EF Toklat trib	DENA	2007	Summer
DG01408	DENA-GRTS-014	EF Toklat trib	DENA	2008	Summer
DG01409	DENA-GRTS-014	EF Toklat trib	DENA	2009	Summer
DG01410	DENA-GRTS-014	EF Toklat trib	DENA	2010	Summer
DG02707S	DENA-GRTS-027	EF Toklat River	DENA	2007	Summer
DG02708S	DENA-GRTS-027	EF Toklat River	DENA	2008	Summer
DG02708F	DENA-GRTS-027	EF Toklat River	DENA	2008	Fall
DG02709S	DENA-GRTS-027	EF Toklat River	DENA	2009	Summer
DG02709F	DENA-GRTS-027	EF Toklat River	DENA	2009	Fall
DG02710S	DENA-GRTS-027	EF Toklat River	DENA	2010	Summer
DG02710F	DENA-GRTS-027	EF Toklat River	DENA	2010	Fall
DG03908	DENA-GRTS-039	upper Moose Creek	DENA	2008	Summer
DG03910	DENA-GRTS-039	upper Moose Creek	DENA	2010	Summer
DG07110	DENA-GRTS-071	Stony Creek	DENA	2010	Summer
DG08407	DENA-GRTS-084	Igloo Creek	DENA	2007	Summer
DG08408	DENA-GRTS-084	Igloo Creek	DENA	2008	Summer
DG08409	DENA-GRTS-084	Igloo Creek	DENA	2009	Summer
DG08410	DENA-GRTS-084	Igloo Creek	DENA	2010	Summer
DG08411	DENA-GRTS-084	Igloo Creek	DENA	2011	Summer
DG10407	DENA-GRTS-104	Sanctuary River	DENA	2007	Summer
DG10408	DENA-GRTS-104	Sanctuary River	DENA	2008	Summer
DG10410	DENA-GRTS-104	Sanctuary River	DENA	2010	Summer
DG12207	DENA-GRTS-122	Highway Pass Creek	DENA	2007	Summer
DG12210	DENA-GRTS-122	Highway Pass Creek	DENA	2010	Summer
DG12211	DENA-GRTS-122	Highway Pass Creek	DENA	2011	Summer
DG12710	DENA-GRTS-127	Birch Creek	DENA	2010	Summer
DG13410	DENA-GRTS-134	Marten Creek	DENA	2010	Summer
DG17310	DENA-GRTS-173	Sanctuary River trib	DENA	2010	Summer
DG18310	DENA-GRTS-183	McLeod Creek	DENA	2010	Summer
W0106	WRST-001	trib W of Tana	WRST	2006	Summer
W0206	WRST-002	Chakina River braid	WRST	2006	Summer
W0306	WRST-003	trib E of Tebay	WRST	2006	Summer
W0406S	WRST-004	Chalk Creek	WRST	2006	Summer
W0406F	WRST-004	Chalk Creek	WRST	2006	Fall
W0407F	WRST-004	Chalk Creek	WRST	2007	Fall
W0408S	WRST-004	Chalk Creek	WRST	2008	Summer
W0408F	WRST-004	Chalk Creek	WRST	2008	Fall
W0409J	WRST-004	Chalk Creek	WRST	2009	Summer
W0409A	WRST-004	Chalk Creek	WRST	2009	Summer

Abbreviation	Original Sitecode	Site Name	Park	Year	Season
W0409F	WRST-004	Chalk Creek	WRST	2009	Fall
W0410J	WRST-004	Chalk Creek	WRST	2010	Summer
W0411S	WRST-004	Chalk Creek	WRST	2011	Summer
W0411F	WRST-004	Chalk Creek	WRST	2011	Fall
W0410A	WRST-004	Chalk Creek	WRST	2010	Summer
W0506	WRST-005	Swift Creek	WRST	2006	Summer
W0606S	WRST-006	Rock Creek WRST	WRST	2006	Summer
W0606F	WRST-006	Rock WRST	WRST	2006	Fall
W0608S	WRST-006	Rock Creek WRST	WRST	2008	Summer
W0608F	WRST-006	Rock Creek	WRST	2008	Fall
W0609S	WRST-006	Rock Creek WRST	WRST	2009	Summer
W0609F	WRST-006	Rock Creek WRST	WRST	2009	Fall
W0610J	WRST-006	Rock Creek WRST	WRST	2010	Summer
W0611S	WRST-006	Rock Creek WRST	WRST	2011	Summer
W0611F	WRST-006	Rock Creek WRST	WRST	2011	Fall
W0610A	WRST-006	Rock Creek WRST	WRST	2010	Summer
W0706	WRST-007	Gravel Creek	WRST	2006	Summer
W0806	WRST-008	Beaver Creek	WRST	2006	Summer
W0808	WRST-008	Beaver Creek	WRST	2008	Summer
W0906	WRST-009	Rock Lake outflow	WRST	2006	Summer
W1006	WRST-010	Ptarmigan Creek	WRST	2006	Summer
W1307	WRST-013	Gilahina River	WRST	2007	Summer
W1309	WRST-013	Gilahina River	WRST	2009	Summer
W1406	WRST-014	Skookum Creek	WRST	2006	Summer
W1506F	WRST-015	Little Jack	WRST	2006	Fall
W1508S	WRST-015	Little Jack	WRST	2008	Summer
W1509S	WRST-015	Little Jack	WRST	2009	Summer
W1511S	WRST-015	Little Jack	WRST	2011	Summer
W1510S	WRST-015	Little Jack	WRST	2010	Summer
W1607	WRST-016	Nizina trib	WRST	2007	Summer
W1707	WRST-017	Lake Creek WRST	WRST	2007	Summer
W1708	WRST-017	Lake Creek WRST	WRST	2008	Summer
W1907S	WRST-019	Willow Creek	WRST	2007	Summer
W1907F	WRST-019	Willow Creek	WRST	2007	Fall
W1908S	WRST-019	Willow Creek	WRST	2008	Summer
W2007	WRST-020	Skolai Pass stream	WRST	2007	Summer
W2107	WRST-021	Moonshine Creek	WRST	2007	Summer
W2207	WRST-022	4th of July Creek	WRST	2007	Summer
W2307	WRST-023	Bear Creek WRST	WRST	2007	Summer
W2407	WRST-024	Amy Creek	WRST	2007	Summer
W2507F	WRST-025	May Creek	WRST	2007	Fall
W2607F	WRST-026	Galiano West creek	WRST	2007	Fall

Abbreviation	Original Sitecode	Site Name	Park	Year	Season
W2707F	WRST-027	Icy Bay stream	WRST	2007	Fall
W2807F	WRST-028	Independence Creek	WRST	2007	Fall
W2907F	WRST-029	Little Esker Creek	WRST	2007	Fall
W3608	WRST-036	Rufus Creek	WRST	2008	Summer
W3609	WRST-036	Rufus Creek	WRST	2009	Summer
W3808	WRST-038	Horsfeld Creek	WRST	2008	Summer
W3908	WRST-039	Caribou Creek	WRST	2008	Summer
W5009	WRST-050	Nadina River	WRST	2009	Summer
W6011	WRST-060	Paco Creek	WRST	2011	Summer
W6111	WRST-061	Tanada Creek	WRST	2011	Summer
WG00408	WRST-GRTS-004	Notch Creek trib	WRST	2008	Summer
WG00610	WRST-GRTS-006	Sanford FP stream	WRST	2010	Summer
WG00809	WRST-GRTS-008	McCarthy Creek	WRST	2009	Summer
WG01208	WRST-GRTS-012	Bryan Creek	WRST	2008	Summer
WG03011F	WRST-GRTS-030	Strelna Creek crossing	WRST	2011	Fall
WG03608	WRST-GRTS-036	Solo Creek trib	WRST	2008	Summer
WG10610	WRST-GRTS-106	Trail Creek	WRST	2010	Summer
WG13006F	WRST-GRTS-130	Jack Creek	WRST	2006	Fall
WG13007F	WRST-GRTS-130	Jack Creek	WRST	2007	Fall
WG13008F	WRST-GRTS-130	Jack Creek	WRST	2008	Fall
WG13009S	WRST-GRTS-130	Jack Creek	WRST	2009	Summer
WG13009F	WRST-GRTS-130	Jack Creek	WRST	2009	Fall
WG13010S	WRST-GRTS-130	Jack Creek	WRST	2010	Summer
WG13011S	WRST-GRTS-130	Jack Creek	WRST	2011	Summer
WG13110	WRST-GRTS-131	Iceberg Bench stream	WRST	2010	Summer
WG20409	WRST-GRTS-204	Tana River trib	WRST	2009	Summer
WG21509	WRST-GRTS-215	Nadina River trib	WRST	2009	Summer
WG22010	WRST-GRTS-220	high elev Tana trib	WRST	2010	Summer
WG22309	WRST-GRTS-223	Sheep Gulch	WRST	2009	Summer
WG22409	WRST-GRTS-224	Clear Stream trib	WRST	2009	Summer
WG22809	WRST-GRTS-228	Hunter Creek	WRST	2009	Summer
WG23411	WRST-GRTS-234	Stuver Creek trib	WRST	2011	Summer
WG24110	WRST-GRTS-241	Mt. Sanford stream	WRST	2010	Summer
WG24711	WRST-GRTS-247	Paco's River	WRST	2011	Summer
WG25111	WRST-GRTS-251	Hanagita River	WRST	2011	Summer
WG28910	WRST-GRTS-289	Sheep Glacier trib	WRST	2010	Summer
WG30208	WRST-GRTS-302	NF White River	WRST	2008	Summer
WG32407	WRST-GRTS-324	Young Creek	WRST	2007	Summer
WG32410	WRST-GRTS-324	Young Creek	WRST	2010	Summer
WG34510	WRST-GRTS-345	Capital Mtn stream	WRST	2010	Summer
WG35510	WRST-GRTS-355	Chokosna River trib	WRST	2010	Summer
WG37511	WRST-GRTS-375	Bridge Creek	WRST	2011	Summer

Abbreviation	Original Sitecode	Site Name	Park	Year	Season
YG00111	YUCH-GRTS-001	upper Charley River	YUCH	2011	Summer
YG00311	YUCH-GRTS-003	Coal Creek	YUCH	2011	Summer
YG03511	YUCH-GRTS-035	Coal Creek @ strip	YUCH	2011	Summer
YG09911	YUCH-GRTS-099	Seventymile River trib	YUCH	2011	Summer

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