



Central Alaska Network Flowing Waters Monitoring Program

2010 Annual Report

Natural Resource Technical Report NPS/CAKN/NRTR—2013/727



ON THE COVER

View towards Iceberg Lake and the Bagley Icefield, Wrangell-St. Elias National Park and Preserve
Photograph by: Trey Simmons

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Abstract

The 2010 field season marked the fifth year of development of the flowing waters portion of the Central Alaska Network (CAKN) Inventory and Monitoring Program, also known as the Vital Signs Program. Data collection occurred in both Wrangell-St. Elias National Park and Preserve (WRST) and in Denali National Park and Preserve (DENA). The purposes of the study were to 1) continue to refine field protocols and logistics related to the collection of relevant data in DENA and WRST streams and rivers; 2) implement a multi-panel sampling design in WRST; 3) begin sampling GRTS-selected synoptic sites in DENA, and 4) implement the long-term flowing water monitoring program. The data collected included biological (benthic macroinvertebrate and diatom samples, fish identity and size, riparian vegetation type), physical (channel geometry, substrate, etc.) and chemical (collected in situ as well as in water samples for later analysis) information. Data were collected during a total of 73 site visits across the two park units; 42 unique sites were sampled. Continuously recording temperature data loggers were deployed at 15 sites across the network. Near-record flooding occurred in late July in the Nabesna area. Very high flows led to the loss of two pressure transducers in Jack Creek, as well as causing substantial channel change in some streams.

In DENA, another four previously unreported macroinvertebrate taxa were documented, bringing the four-year total of newly-documented taxa to 42. Network-wide, the program has now documented 150 unique macroinvertebrate taxa and 440 diatom species. Of these, nine invertebrate taxa and 44 diatom taxa were first reported in 2010. The fourth year of data collected from streams involved in the Long-Term Ecological Monitoring program in DENA confirmed earlier indications that there are incompatibilities between data collected using LTEM methodologies and those collected using CAKN methodologies. Some analyses were conducted on the year-to-year variability of water chemistry and macroinvertebrate data collected since 2006. A substantial portion of the temporal variation in water chemistry, at least for some streams and some constituents, appeared to be attributable to differences in discharge among sampling events. Year-to-year variation in macroinvertebrate composition at most streams appears to be moderate by comparison, and nearly two thirds of the apparent variation is likely due to sampling error. Invertebrate and environmental data collected in 2010 were used to refine the preliminary RIVPACS predictive biological assessment model that was constructed for network streams last year. 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.

Acknowledgements

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

Introduction

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

In 2010, the purposes of the program were to continue to evaluate and refine the existing field methods, maintain the continuity of existing data streams by sampling sites along the DENA park road that were part of the LTEM program and to continue implementation of the multi-panel survey design in Wrangell-St. Elias National Park and Preserve (WRST). In addition, a major focus in 2010 was to begin sampling GRTS-selected synoptic sites in DENA with an eye toward evaluating the logistical feasibility of implementing the multi-panel survey design for the monitoring program in all 3 network parks. The design includes annual sampling at a small number of accessible “sentinel” sites, along with long rotation sampling at a large number of “synoptic” sites that were selected using the generalized random tessellation stratified (GRTS) algorithm, which generates a spatially-balanced, probabilistic sample (Overton and Stehman 1993, Stevens and Olsen 2004). Frequent sampling at sentinel sites will provide sensitivity to trends in various metrics, whereas parkwide inference will be established for each park unit using the synoptic GRTS sites. The ultimate goal for WRST is to sample 10 synoptic sites per year, on an approximately 10-year return interval, for a total of 100 sites. For DENA, the goal is to visit six sites per year on an approximately 10-year return interval, for a total of 60 synoptic sites. For Yukon-Charley National Preserve, the goal is to visit 4 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 the first several years of the program, it is likely that ~20-23 sentinel sites will be in the final set, 10-12 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.

In 2008, the feasibility of using GRTS to select a probabilistic sample of synoptic sites was first tested in WRST. After editing the National Hydrography Dataset (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 and 2010 field seasons. For the most part, the field evaluations have held up when site visits are actually attempted.

A similar approach was taken in 2010 for selecting and evaluating GRTS sites 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 edited out. 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 was the case 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. A list of 240 potential sampling sites was generated for DENA.

The logistical challenges of working in remote Alaskan parklands have forced the CAKN to reconsider the strict application of the GRTS sampling methodology. In traditional GRTS sampling, sites are sampled in numerical order, moving down a list. If a particular site cannot be sampled, it is simply eliminated from the target population, and the next comparable (same accessibility and stream order) site on the list is sampled. However, this approach is unrealistic for CAKN, where consecutive sites may be hundreds of miles apart, and require helicopter access as well. For example, the first GRTS site on the list for WRST, WRST-GRTS-001, is a tributary to Malaspina Lake, or possibly a side channel of the tributary. Accessing this site would require basing a helicopter or floatplane out of Yakutat. 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. 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, far northeast for WRST, south of the Alaska Range, far west 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.

In 2010, the CAKN was able to sample 10 sites from the GRTS list in Wrangell-St. Elias. One of these, WRST-GRTS-130 (Jack Creek), coincides with a road-accessible sentinel site that has been sampled since 2006; the other 9 were new sites and were reached using a helicopter. This

brings the 2-year total to 21 GRTS sites (including Jack Creek), suggesting that the target of 10 GRTS sites per year is feasible, at least as an average (12 unique GRTS sites were sampled in 2009). In addition, eight sentinel sites along the road system in WRST were sampled (nine if Jack Creek is included). This suggests that the two-tiered survey design outlined above is feasible, although adding GRTS and sentinel sites in YUCH in 2011 may prove challenging. Table 1 provides a list of sites sampled in 2010.

Invertebrate and environmental data collected from 2010 were used to improve the performance of the RIVPACS (or O/E) combined with data from earlier years to develop and evaluate RIVPACS-type biological assessment models. These methods use natural environmental gradients to predict the invertebrate community that would be expected at a given site in the absence of anthropogenic stress. Deviations from the expected community composition serve as a metric of ecosystem impairment. As such, they have the potential to allow the CAKN to evaluate contemporaneous ecological integrity in network streams as well as detect changes in otherwise pristine streams due to the effects of remote stressors such as climate change.

Methods

Study Area

Data were collected during a total of 73 site visits at 42 unique sites in 2010. In DENA there were 33 site visits to 18 unique sites, and in WRST there were 40 visits to 24 unique sites (Table 1). The full suite of data was not collected during all visits, however. The locations of the sites sampled are shown in Figure 1. Most of the DENA sites were located along or near the park road. These sites included 10 sites that have been sampled since 1994 as part of the LTEM program. Twelve of the sites had been sampled by the CAKN at least once during 2007-2009; Stony Creek trib was sampled for the first time in 2010. Four remote GRTS sites were sampled in DENA in 2010. In WRST, 10 sites along the road system were sampled, seven of which had been sampled in previous years. The new road sites were all from the GRTS list, and included Tanada Creek, Caribou Creek and a small tributary of Jack Creek (the latter two sites were both dry at the time of the sampling visit, although flowing earlier in the year). Nine remote sites, all of them from the GRTS list, were also sampled by helicopter.

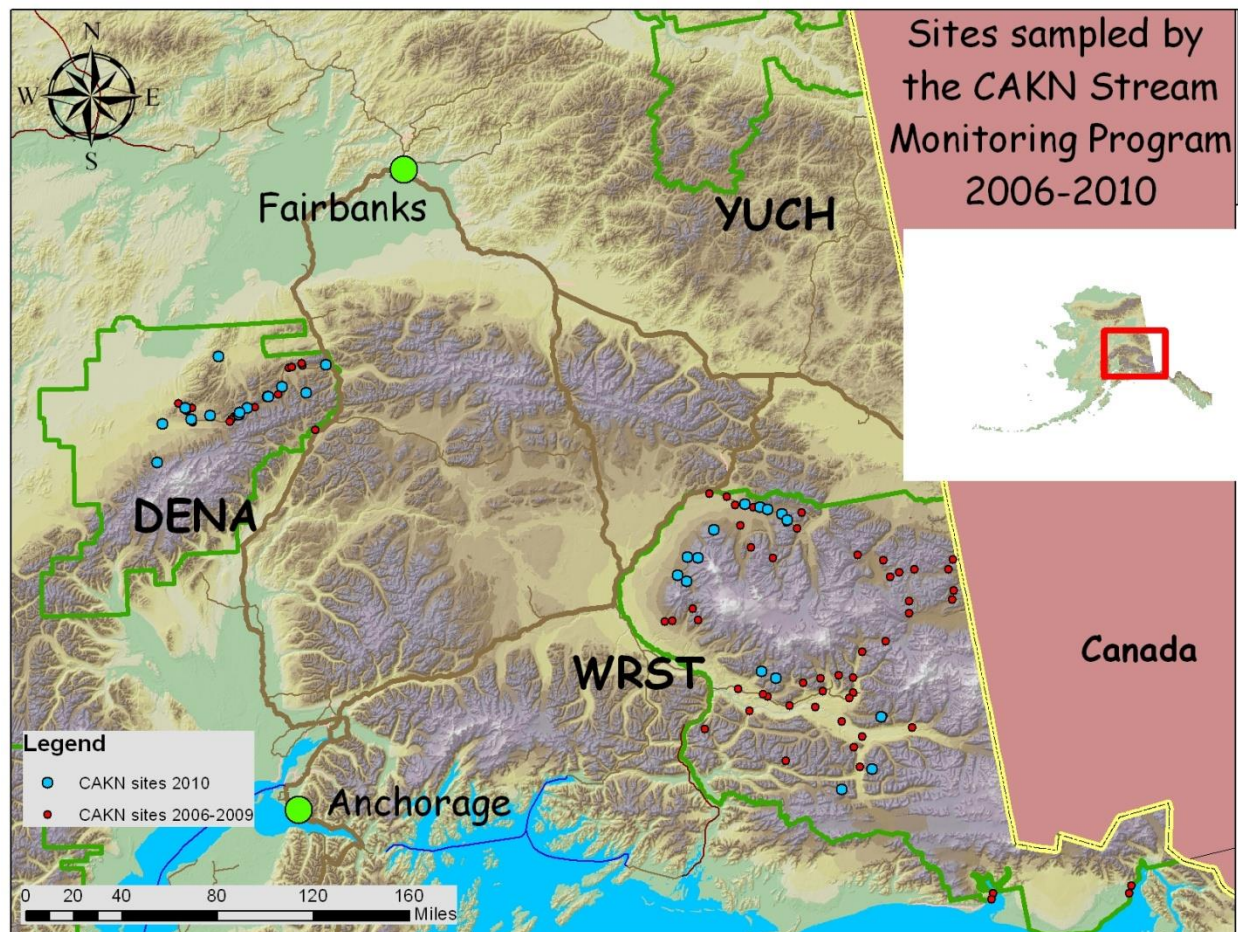


Figure 1. Locations of 2010 CAKN stream monitoring sites (shown in blue). Sites sampled in previous years are shown in red. Note that only sites where biological samples were collected are shown.

Table 1. List of sites sampled in 2010 with location and brief description. A * indicates sites that have been sampled at least once in previous years. A *b* indicates sites where biological data were collected.

Site	Park	Description
Rock Creek ^{*b}	DENA	high-gradient, confined-channel step-pool stream (LTEM site)
Sanctuary River [*]	DENA	large glacially-influenced river (LTEM site)
Moose Creek at bridge ^{*b}	DENA	large plane bed stream in Kantishna Hills (LTEM site)
Highway Pass Creek ^{*b}	DENA	unconsolidated braided stream (LTEM site)
McKinley Bar Trail Creek ^{*b}	DENA	spring-fed mossy meadow stream
Igloo Creek ^{*b}	DENA	large forested stream (LTEM site)
Tattler Creek [*]	DENA	small steep braided stream (LTEM site)
E.F. Toklat River ^{*b}	DENA	large turbid braided glacial river (LTEM site)
Hogan Creek [*]	DENA	small groundwater-fed stream (LTEM site)
Savage River [*]	DENA	large river with some glacial influence (LTEM site)
E.F. Toklat tributary ^{*b}	DENA	clearwater tributary to E.F. Toklat River (LTEM site)
Upper Moose Creek ^{*b}	DENA	large plane-bed stream in Kantishna Hills
Stony Creek ^{*b}	DENA	steep step-pool unconfined channel stream
Stony Creek tributary ^b	DENA	very small stream on Highway Pass
Birch Creek ^b	DENA	steep turbid glacial stream
Marten Creek ^b	DENA	forested pool-riffle stream
Sanctuary River tributary ^b	DENA	high-elevation tundra stream
McLeod Creek ^b	DENA	low-elevation sandy-bottomed stream
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
Jack Creek at bridge ^{*b}	WRST	large lake outlet stream along Nabesna Road
Little Jack Creek [*]	WRST	clearwater stream along Nabesna Road
Gilahina River [*]	WRST	large clearwater stream along the McCarthy Road
Young Creek ^{*b}	WRST	large alpine tundra pool-riffle stream
Rufus Creek [*]	WRST	stable groundwater stream along Nabesna Road
Jack Creek tributary [*]	WRST	very steep unconsolidated intermittent stream

Table 1. List of sites sampled in 2010 with location and brief description. A * indicates sites that have been sampled at least once in previous years. A *b* indicates sites where biological data were collected. (continued)

Site	Park	Description
Caribou Creek*	WRST	intermittent stream along Nabesna Road (2 sites sampled)
Long Lake Creek*	WRST	low-gradient forested lake outlet stream
Mile 27 Stream	WRST	narrow deep meadow stream
Sanford River floodplain stream ^b	WRST	small stable mossy stream on Sanford River floodplain
Crystal Creek	WRST	small steep forested lake outlet with beaver dams
Black Mountain East stream	WRST	mostly sand/gravel substrate, possibly intermittent
Trail Creek ^b	WRST	shallow clear stream on Kuskulana floodplain
Iceberg Bench stream ^b	WRST	small spring-fed stream in alpine basin
Sanford River trib ^b	WRST	intermittent stream in tundra
Black Mountain East stream	WRST	deep meadow stream
High elevation Tana trib ^b	WRST	steep alpine tundra stream
Mt Sanford stream ^b	WRST	steep alpine tundra stream
Sheep Glacier stream tributary ^b	WRST	small steep unconsolidated alpine stream
Capital Mountain stream ^b	WRST	steep rocky alpine stream
Chokosna River tributary ^b	WRST	small alpine tundra stream

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 4-digit hydrographic units that encompass WRST or DENA were first merged, and then clipped to the park boundary using ArcGIS Desktop 9.3 (ESRI, Redlands, CA). The RivEX network analysis tool (Hornby 2009) was used to evaluate the NHD for topological errors (*e.g.*, disconnected polylines). Corrections to the network were effected using tools available in ArcGIS. The RivEX tool was then used with the edited and corrected NHD layer to attribute each stream segment with Strahler stream order (Strahler 1952). Stream order can be used as a surrogate for stream size; because the program is largely limited to sampling wadeable streams, it was important to 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. The idea was that these corridors would provide access either by boat or by fixed-wing aircraft (due to the likely presence of large gravel bars that might serve as landing strips); however, in most cases it turned out that use of a helicopter would be required to access these sites. For DENA, the accessibility surface included only the Park Road, trails and floatplane-accessible lakes.

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 deemed likely that the majority of sites would not be sampleable, making it difficult to identify a set of 100 sampleable sites without a substantially longer starting list of possibilities. This has turned out to be accurate. The unequal weighting function was used to over-represent 2nd and 3rd order stream segments in the site list. 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). All 640 sites (400 for WRST, 240 for DENA) were then examined using a variety of remotely sensed and GIS data to determine whether they were part of the target population. Field evaluation of the sites determined to be part of the target population is ongoing.

Reach definition

Sampling reaches were defined using guidelines from the Environmental Protection Agency (EPA) 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), the loss of accuracy is minimal because the goal of the program is to characterize the reach, rather than to establish monumented cross sections; therefore, the exact locations at which data are collected should not matter.

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. 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. This sample represents a total of 0.72 m^2 of streambed. At some sites, this methodology was modified as follows. For “occupancy”-type samples (quasi-replicated samples that are designed to be used in the development of occupancy models), each adjacent pair of 0.09 m^2 subsamples was composited, for a total of four quasi-replicated samples, each representing 0.18 m^2 of streambed. Macroinvertebrates and organic detritus were separated from cobble and gravel and preserved in 70% ethanol. Macroinvertebrates were sorted and identified to the lowest practical taxonomic level, generally genus, by Mike Cole, a taxonomist for ABR, Inc. Due to variability within and among samples in the taxonomic resolution that could be achieved, the CAKN has established a set of Operational Taxonomic Units (OTU) to assure that taxa are not double (or triple) counted (Simmons, unpublished). In some cases this required deleting coarse-resolution data (generally family level); in other cases, higher-resolution data (species or genus) were collapsed back to coarser resolution (genus, sub-family or family). The net result, often referred to as OTU richness, is therefore a conservative estimate of the true richness at a given site.



Figure 2. 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 2010 to collect fish data; this Vital Sign is currently being re-evaluated.

Physical and chemical data collection

We used a YSI QS650 sonde to collect temperature, specific conductivity, pH and dissolved oxygen *in situ*. Data were collected in riffles or runs and generally in midstream at the bottom of the reach. In addition, water chemistry samples were collected for later laboratory analysis. The samples were collected at the same location as the *in situ* water chemistry. A 1-l bottle was rinsed three times with stream water, and then used to collect the main sample. A 500-mL

unfiltered sample for total nitrogen and total phosphorous was transferred into an acid-washed Nalgene sample bottle and placed in a cooler with dry ice to freeze. The other 500 mL was filtered through a 0.45 μm filter (previously rinsed with 50 mL of stream water). A 250-mL aliquot (for nitrate, ammonium, phosphate and common ions) was transferred to an acid-washed Nalgene bottle and placed in a cooler with dry ice to freeze. A 125-mL aliquot of filtrate (for dissolved organic carbon and silicon analysis) was transferred to an acid-washed Nalgene bottle and kept on ice. In some cases it was not possible to freeze the samples in the field; in these cases, the samples were kept cool in stream water and frozen as rapidly as possible (generally within an hour or two). Analytical water chemistry data were provided by the Cooperative



Figure 3. Filtering water chemistry samples in the field (Independence Creek, WRST).

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 the Environmental and Natural Resources Institute at the University of Alaska Anchorage.

Physical data collection protocols were largely based on EMAP WSA protocols (USEPA 2004). At each transect, we measured depth (five measurements), width (wetted and bankfull), channel height (bankfull and incised), undercut banks, canopy cover (six measurements) and substrate size class (using a gravelometer – five measurements at depth locations). In intertransect

segments, we measured thalweg depth and habitat type (ten measurements), width (one measurement), substrate (five cobbles along width measurement cross-section), woody debris (by size class) and fish cover (macrophytes, filamentous algae, boulders and undercut banks (qualitative estimate of extent). We measured reach slope using a transit level at some sites. We also measured discharge at a subset of sites using a Marsh-McBirney flowmeter and a topsetting wading rod.

Continuous data collection

Temperature-recording dataloggers were installed in 10 streams in May and early June (Table 2). In addition, two continuously recording pressure transducer/temperature recorders were installed at the Jack Creek site (WRST), and one pressure transducer/temperature recorder was deployed at Caribou Creek in conjunction with the temperature datalogger. Dataloggers could not be deployed to most DENA sites due to excessive instream ice remaining at the time of the spring deployment trip. Dataloggers were generally secured using vinyl-coated steel cable to attach them to riparian trees. Loggers were additionally wired to the underside of large anchor rocks to keep them in place. Where possible, loggers were placed in relatively deep, fast-flowing water (*e.g.*, in a run). Care was taken to minimize the visibility of both the logger and the anchoring system. Three other temperature loggers, both installed at remote sites in 2008, could not be retrieved before the end of the 2010 field season, and remain in place, or so it is hoped. Loggers were set to record temperature every 60 minutes. This resulted in an approximately 3-year maximum lifespan for the data record. The Jack Creek pressure transducers were lost in the record flood event in late July.

Table 2. List of sites where temperature loggers were installed in 2010, along with brief site descriptions.

Site	Park	Description
Tattler Creek	DENA	small steep braided stream (LTEM site)
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
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

Data Analysis

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

Results and Discussion

Water Chemistry

The water chemistry results from 2010 (Table 3) revealed a very wide range of conditions across the two parks, consistent with results from previous years. Descriptive statistics derived from all of the water chemistry data collected from 2006-2010 are very similar to those calculated in 2010 alone. In general the maximum and minimum values for a given constituent vary from year to year much more than the mean and median values, as would be expected. With five years of water chemistry data in hand, it is now possible to start looking at spatial and temporal patterns in some of the constituents. Many chemical constituents vary over several orders of magnitude across CAKN. For example, specific conductivity varies from a low of 10 $\mu\text{S}/\text{cm}$ (Iceberg Bench stream in WRST) to a high of 1295 $\mu\text{S}/\text{cm}$ (Stony Creek in DENA), a nearly 130-fold range. Sulfate is even more variable, ranging from 0.14 mg/L (tributary of Rock Creek in WRST) to over 143 mg/L (Rock Creek in WRST), greater than a 1000-fold range. Even more remarkable is that these two sampling sites are less than 5 meters apart.

Table 3. Summary water chemistry statistics from 2010 samples.

Parameter	Minimum	Mean	Median	Maximum
Temperature (instantaneous)	-0.05°C	5.1°C	5.7°C	17.9°C
Specific conductance	10 $\mu\text{S}/\text{cm}$	366 $\mu\text{S}/\text{cm}$	341 $\mu\text{S}/\text{cm}$	1295 $\mu\text{S}/\text{cm}$
pH	7.57	8.11	8.12	8.56
Alkalinity (HCO_3^- -C)	3.25 mg/L	108.8 mg/L	110.2 mg/L	318.2 mg/L
Total nitrogen	20 $\mu\text{g}/\text{L}$	168 $\mu\text{g}/\text{L}$	150 $\mu\text{g}/\text{L}$	350 $\mu\text{g}/\text{L}$
Nitrate-nitrogen	<1 $\mu\text{g}/\text{L}$	97 $\mu\text{g}/\text{L}$	79 $\mu\text{g}/\text{L}$	279 $\mu\text{g}/\text{L}$
Dissolved organic nitrogen	0 $\mu\text{g}/\text{L}$	53 $\mu\text{g}/\text{L}$	43 $\mu\text{g}/\text{L}$	240 $\mu\text{g}/\text{L}$
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 $\mu\text{g}/\text{L}$	2.7 $\mu\text{g}/\text{L}$ *	1 $\mu\text{g}/\text{L}$ *	35 $\mu\text{g}/\text{L}$
Sulfate-sulfur	0.14 mg/L	24.8 mg/L	17.2 mg/L	143 mg/L
Dissolved organic carbon	0.24 mg/L	1.6 mg/L	1.2 mg/L	9 mg/L

*70% of samples tested in 2010 had phosphate-P levels at or below the level of detectability reported by the Cooperative Chemical Analytical Laboratory (CCAL). I substituted values of $\frac{1}{2}$ the detection limit of 1 $\mu\text{g}/\text{L}$ for calculating descriptive statistics, although with the understanding that it may introduce bias (Helsel 2006).

Another pattern that was confirmed with the 2010 data is the generally low levels of phosphate-P in CAKN streams. The mean concentration of phosphate-P among all samples collected from 2006-2010 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 these streams (24 $\mu\text{g}/\text{L}$) is substantially 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 12 $\mu\text{g}/\text{L}$, which is strikingly low. In fact, only a handful of streams in the CAKN had a TP

concentration higher than 50 $\mu\text{g/L}$ – most of these are glacial streams with high sediment loads and very low levels of dissolved phosphate. This suggests that suspended sediment was the source of the high TP levels and that much of it is not biologically available.

The generally low phosphate levels in CAKN streams are reflected in N:P ratios, which give an indication of nutrient limitations in aquatic ecosystems. In general, N:P ratios greater than 16 (the “Redfield Ratio”, 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 182:1, and the range goes as high as 2000:1 (Swift Creek in WRST). Fully 90% of the samples collected have N:P ratios greater than 16. 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 phosphate-P. Because for most samples, phosphate-P was near the limit of detection, the variation in N:P, both within and among sites, is probably exaggerated somewhat. Interestingly, all sites in DENA had N:P ratios above the Redfield Ratio, and the mean value for DENA samples (361) was substantially higher than for the dataset as a whole.

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 161 samples so far analyzed, 63% are below the level of detection for SRP ($1\mu\text{g/L}$) and 66% are below the level of detection for ammonium nitrogen ($10\mu\text{g/L}$). 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, the CAKN began requesting TDN analysis in 2009. However, the TDN results were problematic in that for most samples, TDN was higher than total nitrogen (TN), suggesting that the filtered samples were contaminated. Further

investigation revealed that the probable source of the contamination was the nitrocellulose filters themselves. In 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 2010 CAKN samples are in line with expectations based on the literature, with DON comprising a substantial proportion of total dissolved nitrogen (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 in 2010 CAKN streams (n = 38).

	Concentration ($\mu\text{g/L}$)	% of Total Dissolved N
Minimum	0	0
Mean	54	34%
Median	43	25%
Maximum	240	89%

Macroinvertebrates

A total of 31 macroinvertebrate samples were collected from 15 DENA streams (16 site visits) and 13 WRST streams (15 site visits) in 2010. As was the case in previous years, observed OTU taxa richness varied tremendously among the sampled sites, ranging from five unique taxa at glacial Birch Creek in DENA to 34 at both Moose Creek sites sampled in DENA. The mean richness across all samples collected in 2010 was 20.9 taxa, which is similar to the mean richness observed in previous years (range 17.9 – 22.6). Chironomid midges accounted for approximately 35% of OTU taxa richness across all sites, and 38% of individuals. This is substantially lower than reported by Oswood (1989), who found that chironomids constituted on average 59% of individuals in Alaskan streams, but is similar to the results observed in CAKN streams since 2006 (mean % of richness as chironomids has varied from 30-37%). Densities varied from a low of 33 individuals/ m^2 at Birch Creek in DENA to a high of nearly 41,000 per m^2 at the Sanford River floodplain stream in WRST, with a mean of 2873 individuals/ m^2 , also similar to what has been observed in previous years.

Nine taxa not previously documented were collected in 2010 (Table 5), including the first dytiscid beetle and first amphipod I have observed. This brings the total number of unique taxa observed since 2006 to 150. Between 2007 and 2009 a total of 38 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). In 2010 we collected an additional five previously unreported invertebrate taxa (Table 5), including two new orders (Coleoptera and Amphipoda). This brings the 4-year total to a minimum of 43 newly reported aquatic macroinvertebrate taxa, which is remarkable given the limited geographic and temporal scope of the CAKN program in DENA and reinforces the argument, made following the 2008 field season, that the streams and rivers of DENA remain woefully understudied. One of these newly reported taxa, the stonefly genus *Ostrocerca*, has not been previously reported from anywhere in Alaska and so likely represents a new state record. A similar situation no doubt entails in WRST, but there is no comparable literature available for comparison in that park unit.

Table 5. List of new aquatic invertebrate taxa identified from 2010 CAKN samples. A * indicates taxa collected in DENA that have not previously been reported (Conn 1998, Milner et al. 2003).

Plecoptera (stoneflies)

*Zapada columbiana**
Alaskaperla ovibovis
Ostrocerca sp.*

Ephemeroptera (mayflies)

Acentrella feropagus

Coleoptera (beetles)

Hydroporinae (a subfamily of *Dytiscidae*)*

Diptera (true flies)

Stegopterna sp.
Boreoheptagyia sp.
Parachironomus sp.*

Amphipoda (amphipod crustaceans)

Stygobromus sp.*

A total of 150 unique macroinvertebrate taxa have now been identified in the five years of the CAKN program. It is likely that we will continue to add to this total for a number of years given the relative paucity of stream macroinvertebrate data from these parks, particularly from WRST. Three of these taxa have never been formally identified, and would represent new species, or potentially new genera. One of these unidentified taxa has been collected at multiple sites since 2007. This previously unidentified midge of the subfamily Orthocladiinae, first identified in samples from 3 sites in 2007, and collected again at 4 sites in 2008 and 4 sites in 2009, was also collected at 2 sites in 2010. The midge has been collected in all 3 years of sampling (2007-2009) from Hogan Creek (DENA), and in both 2008 and 2009 in the Travertine Stream (DENA). In 2009 it was also collected in Little Stony Creek (DENA), in both the summer and fall samples, but was not found in any samples from WRST. In contrast, in 2010 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 or 2010. Lastly, a previously unidentified member of the family Empididae (dance flies) was collected at 7 sites in 2009. It is likely that this taxon was collected in previous years as well, but was reported as *Neoplata*, 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 2010, this taxon was identified in 14 samples (13 sites) from both DENA and WRST. Also in 2010, an attempt was made to field preserve specimens for DNA analysis. This effort will continue in 2011, and it should be possible to identify this taxon in the near future.

Benthic Diatoms

In 2010, 31 diatom samples were collected from 28 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 one species (Birch Creek in DENA) to a high of 54 species (Stony Creek tributary in DENA), with a mean of 30 species per site. A total of 441 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, and another 45 in 2010. 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 has consistently supported 12-22 species of diatoms during the summer, as opposed to only three to five macroinvertebrate taxa. Densities varied from a low of only 671 cells/cm² at glacial Birch Creek in DENA to a high of 3.4×10^7 cells/cm² at the small, stable Sanford River floodplain stream in WRST.

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 2010, *Didymosphenia* was identified at four of the 28 sites sampled. All of these sites were located along the Park road in DENA (Sanctuary River, Igloo Creek, EF Toklat tributary and Moose Creek). These findings are consistent with those from previous years. A nuisance bloom was observed in the East Fork Toklat tributary in DENA in 2009; a sample of the bloom material was collected and submitted to Dan Bogan and Dan Rinella at the University of Alaska Anchorage, who confirmed its identity as *D. geminata*. Interestingly, this site and the site where a nuisance bloom was observed in 2008 are in adjacent basins. 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. 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

2010 marked the third year in which CAKN has sampled a glacial river in both the summer and the fall. Glacially-influenced rivers are typically turbid and highly dynamic during the summer melt season, but clear up and stabilize in the fall. This clearwater phase, which occurs in both fall and spring, may provide a critical subsidy to stream consumers, including fish, as the stable hydrology and clear conditions should allow for a burst of primary productivity during the time period after the end of the glacial melt season, but before the angle of the sun, temperature and day length decrease too much (and *vice versa* in the spring). Because climate warming may alter these processes, the CAKN monitoring program is determined to include multi-season monitoring of glacial rivers as an important component. The East Fork of the Toklat River in DENA was chosen as the first glacial river to be monitored in this way. In 2007-2010, 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, and are shown in 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), whereas 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. 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 should provide an invaluable starting point for the determination of long-term trends in the ecology of DENA streams; however, differences in the field and laboratory methods used by 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 by both groups at 10 sites along the park road in 2007 revealed substantial differences in the results produced by the two programs, in both reported taxa richness and community composition, at these 10 sites. Four of these sites have been resampled in all three years from 2008-2010, three 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.

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. This effect should be more pronounced in larger streams, as was observed. The 2008-2010 CAKN data continue to support this idea in general. On average, the mean (2007-2010) observed richness in

Table 7. LTEM sites that were resampled by CAKN in 2007-2010.

Stream name (reach length in meters)	Description of CAKN sampling reach	Years sampled
Hogan Creek (150 m)	Reach starts ~50 m above road	2007, 2008, 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
Igloo Creek (250 m)	Reach starts ~100 m above bridge	2007-2010
Tattler Creek (150 m)	Reach starts ~50 m above road	2007, 2008
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, 2008, 2009
Moose Creek (500 m)	Top of reach at Kantishna bridge	2007-2010
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.

The CAKN samples is 71% 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 the 2007 LTEM samples. 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. 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.

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

Table 8. Comparison of observed adjusted taxa richness (S) and density at LTEM sites in DENA 2007-2010. The 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	19.7	203	331
Igloo Creek	11	18.8	431	2105
Tattler Creek	10	18	517	1107
East Fork Toklat River	3	4.8	19	24
East Fork Toklat tributary	8	19	263	1618
Little Stony Creek	10	12.8	456	4912
Moose Creek	18	27.5	1333	1721
Savage River	13	21	374	644
Highway Pass Creek	5	12.5	127	510
Mean across all samples	10.2	16.7	727	1945

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 2008-2010 are included in the analysis, it was found that the mean Bray-Curtis distance between CAKN stream-year pairs (0.36) was less than the mean distance between LTEM and CAKN samples collected at those 8 streams only a few days apart in 2007 (0.40). Distances between the LTEM samples and both the 2008 (0.49) and 2009 (0.37) CAKN samples were similar to the 2007 results. Most of this difference was again driven by the substantial dissimilarity in samples collected in larger rivers. In contrast, in 2009 the CAKN program collected two samples from Chalk Creek 17 days apart; the Bray-Curtis dissimilarity between these two samples was only 0.17, approximately the same as the dissimilarities between both sets of same-day replicates mentioned above. The substantially larger dissimilarities between the LTEM and CAKN samples suggests there is a systematic problem, at least with respect to larger streams and rivers, in comparing data collected by the two programs.

Initial analyses of year-to-year variability

Water Chemistry

2010 was the fifth year of data collection (fourth for most sites), which provides enough data to begin to look at 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 32 sites with data from at least two years, and 20 sites with data from at least three years. For laboratory water chemistry data, there are 27 sites with data from at least two years, 14 sites with data from at least three years, 11 sites with data from four years and three sites with data from all five years. In addition, a number of sites have data recorded multiple times per year. For laboratory-derived water chemistry, there are 29 sites with data from at least two years, 15 sites with data from at least three years, nine sites with data from four years and three sites with data from all five years. 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. 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
pH	.014	.05	.03
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

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

Table 10. Visit-to-visit variability in water chemistry parameters at sites sampled at least once per year from 2006-2010.

Stream	Parameter	N	Mean	95% CI	CV
Rock Creek	Specific conductance	11	363 μ S/cm	+/- 44	0.21
Chalk Creek	Specific conductance	13	444 μ S/cm	+/- 26	0.16
Jack Creek	Specific conductance	13	391 μ S/cm	+/- 30	0.11
Rock Creek	pH	12	7.91	+/- 0.16	0.04
Chalk Creek	pH	14	8.24	+/- 0.12	0.03
Jack Creek	pH	13	7.94	+/- 0.14	0.03
Rock Creek	Total alkalinity	9	150 mg/L	+/- 13	0.15
Chalk Creek	Total alkalinity	11	157 mg/L	+/- 12	0.13
Jack Creek	Total alkalinity	9	135 mg/L	+/- 11	0.13
Rock Creek	Total nitrogen	9	197 μ g/L	+/- 33	0.25
Chalk Creek	Total nitrogen	11	259 μ g/L	+/- 29	0.19
Jack Creek	Total nitrogen	9	226 μ g/L	+/- 19	0.13
Rock Creek	Dissolved organic carbon	9	3.74 mg/L	+/- 0.60	0.24
Chalk Creek	Dissolved organic carbon	11	1.71 mg/L	+/- 0.55	0.54
Jack Creek	Dissolved organic carbon	9	2.03 mg/L	+/- 0.54	0.40
Rock Creek	Total phosphorous	9	10 μ g/L	+/- 4	0.61
Chalk Creek	Total phosphorous	11	17 μ g/L	+/- 3	0.33
Jack Creek	Total phosphorous	9	18 μ g/L	+/- 8	0.69
Rock Creek	Nitrate-nitrogen	9	64 μ g/L	+/- 16	0.38
Chalk Creek	Nitrate-nitrogen	11	166 μ g/L	+/- 25	0.26
Jack Creek	Nitrate-nitrogen	9	131 μ g/L	+/- 24	0.27
Rock Creek	Sulfate-sulfur	9	17.6 mg/L	+/- 2.5	0.22
Chalk Creek	Sulfate-sulfur	11	24.8 mg/L	+/- 2.2	0.15
Jack Creek	Sulfate-sulfur	9	24.3 mg/L	+/- 1.5	0.09

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.

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. Although the CAKN program does not routinely quantify discharge, 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

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)
Little Stony Creek	.438	ND	.085
Tattler Creek	.376	.105	.104
Hogan Creek	.230	.006	.120
Savage River	.070	ND	.026
Moose Creek	.072	ND	.056
EF Toklat tributary	.357	.044	.096
EF Toklat River	.110	.072	.042
Stony Creek	.593	ND	.137
Igloo Creek	.332	.036	.082
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.

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).

Macroinvertebrates

A preliminary analysis of patterns in macroinvertebrate density and community composition over the five-year sampling period was also conducted. Briefly, abundance data were log transformed to reduce the effects of extreme variations, and a Bray-Curtis dissimilarity matrix was calculated. The matrix itself was examined to assess year-to-year variation in composition at

sites with multiple years of data (Table 12); in addition, cluster analysis and non-metric multidimensional scaling (NMDS) ordination were used to look at overall patterns among sites and across years. The site with the most invertebrate samples is Chalk Creek, where samples have been collected in all five years, including both summer and fall samples. Furthermore, in 2006, five replicate macroinvertebrate samples were collected at Chalk Creek in a single day. The mean between-sample dissimilarity for these five samples was 0.19, meaning that, on average, any two replicates shared 81% of their taxa in common. In August, 2009 a similar procedure was conducted at Little Stony Creek in DENA. Amongst the four replicate samples collected there, mean Bray-Curtis dissimilarity was 0.18. Assuming that these two streams are fairly representative, this can be interpreted to mean that a combination of spatial heterogeneity in the distribution of macroinvertebrates and subsampling error (see *e.g.*, Cao *et al.* 2002, Ostermiller and Hawkins 2004) limits the similarity between any two samples to approximately 80%; in other words, the sampling error averages just less than 20%.

Table 12. Year-to-year variation in macroinvertebrate community composition. Figures given are Bray-Curtis dissimilarities and are averages among all year pairs (e.g., 2006-2007, 2006-2008, 2006-2009, 2006-2010, 2007-2008, 2007-2009, 2007-2010, 2008-2009, 2008-2010 and 2009-2010 if data were available for all 5 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 (14)	.30	.25	.32	.34
Rock Creek WRST (8)	.37	.39	.39	.36
EF Toklat River (7)	.72*	.55*	.67*	.56*
Jack Creek (6)	.33	.38	.40	.41
Little Stony Creek (4)	.29	--	.33	.26
Moose Creek (4)	.30	--	.30	--
EF Toklat tributary (4)	.44	--	.44	--
Igloo Creek (4)	.43	--	.43	--
Little Jack Creek (4)	.33	--	.46	--
McKinley Bar Trail creek (3)	.29	--	.29	--
Willow Creek (3)	.55	--	.60	.65
Hogan Creek (3)	.35	--	.35	--
Sanctuary River (3)	.40	--	.40	--
Savage River (2)	.29	--	.29	--
Beaver Creek (2)	.36	--	.36	--
Lake Creek WRST (2)	.41	--	.41	--
Gilahina River (2)	.32	--	.32	--

Table 12. Year-to-year variation in macroinvertebrate community composition. Figures given are Bray-Curtis dissimilarities and are averages among all year pairs (e.g., 2006-2007, 2006-2008, 2006-2009, 2006-2010, 2007-2008, 2007-2009, 2007-2010, 2008-2009, 2008-2010 and 2009-2010 if data were available for all 5 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)	.35	--	.35	--
McKinley Bar spring (2)	.37	--	.37	--
Upper Igloo Creek (2)	.42	--	.42	--
Tattler Creek (2)	.27	--	.27	--
Upper Moose Creek (2)	.49	--	.49	--
Young Creek (2)	.58	--	.58	--

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

Mean dissimilarity among all Chalk Creek samples was 0.32; however, samples were collected in both summer and fall so this figure includes seasonal as well as interannual variability and variability due to sampling error. Mean dissimilarity between summer-only samples was 0.30, while the mean dissimilarity among fall-only samples was 0.25. Given the sampling error of about 0.20, this suggests that variation in community composition is actually rather low from year to year. Similar results were obtained at Little Stony Creek (Table 12). Furthermore, in 2009 summer samples were collected from Chalk Creek in both July and August. The dissimilarity between these samples was 0.17, or about the same as the average sampling error, suggesting that community composition may not have changed at all over that time. Although it is clearly risky to draw broad conclusions based on only 5 years of data at only 2 sites, these results suggest that a reduction in sampling error could have a substantial impact on the power to detect changes in species composition.

Among all 23 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 0.37. Year-to-year dissimilarity among fall samples was also relatively low at 0.34. There was a substantial degree of variation among streams, however (Table 12), with interannual dissimilarities ranging from a low of 0.27 (Tattler Creek in DENA) to a high of 0.58 (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 = 0.40) 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 4) provides another and in some ways more informative way to evaluate the dissimilarity matrix, because dissimilarities are considered in a relative context. Flexible- β UPGMA (Unweighted Pair Group Method with Arithmetic mean) clustering is an agglomerative hierarchical clustering algorithm, and has a number of recommended properties as an analytical

method for evaluating community data (McCune and Grace 2002). For this analysis, 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 taxa in CAKN streams are rare according to this definition (54 of 125). This is reflective of the high beta-diversity among CAKN stream communities. Of the 125 unique taxa in the data set, only 11 are found at more than 50% of sites, and only 52 are found at more than 10% of the sites. 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 4. 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. Inspection of Figure 4 shows that in most cases, samples from a given site collected in different years cluster together, even if the dendrogram is trimmed with most of the information 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 other instances, samples from consecutive years ended up quite “far” from each other in the dendrogram; that is, on opposite sides of a major split that occurs near the very top of the dendrogram. In some of these cases there is a reasonable environmental explanation, though no way of confirming it. Disturbance may explain the observed difference between 2007, 2008 and 2009 in the E.F. Toklat tributary samples (D2307, D2308 and D2309). The 2007 and 2009 samples are very similar and cluster together, whereas the 2008 sample is quite different. This stream is subject to aufeis events of varying magnitude every spring; substantial aufeis events are known to constitute a significant ecological disturbance and can lead to channel evulsions of varying duration. A similar pattern was observed in the Igloo Creek samples (D1607, D1608, D1609). Igloo Creek and the E.F. Toklat tributary are in adjacent basins and Igloo Creek is also subject to aufeis. Although variation in the intensity and duration of aufeis is an attractive explanation, we do not have the data to confirm or reject it.

The cluster dendrogram also reveals some broad ecological patterns in the distribution of stream biota that reflect underlying physical differences among streams. One important note is that there is little or no underlying geographical pattern; that is, the clusters reflect ecological similarities rather than physical proximity. This is best illustrated by considering lake outlet streams. Lake Creek, the outlet stream of Wonder Lake, is in DENA, but is most closely related in terms of

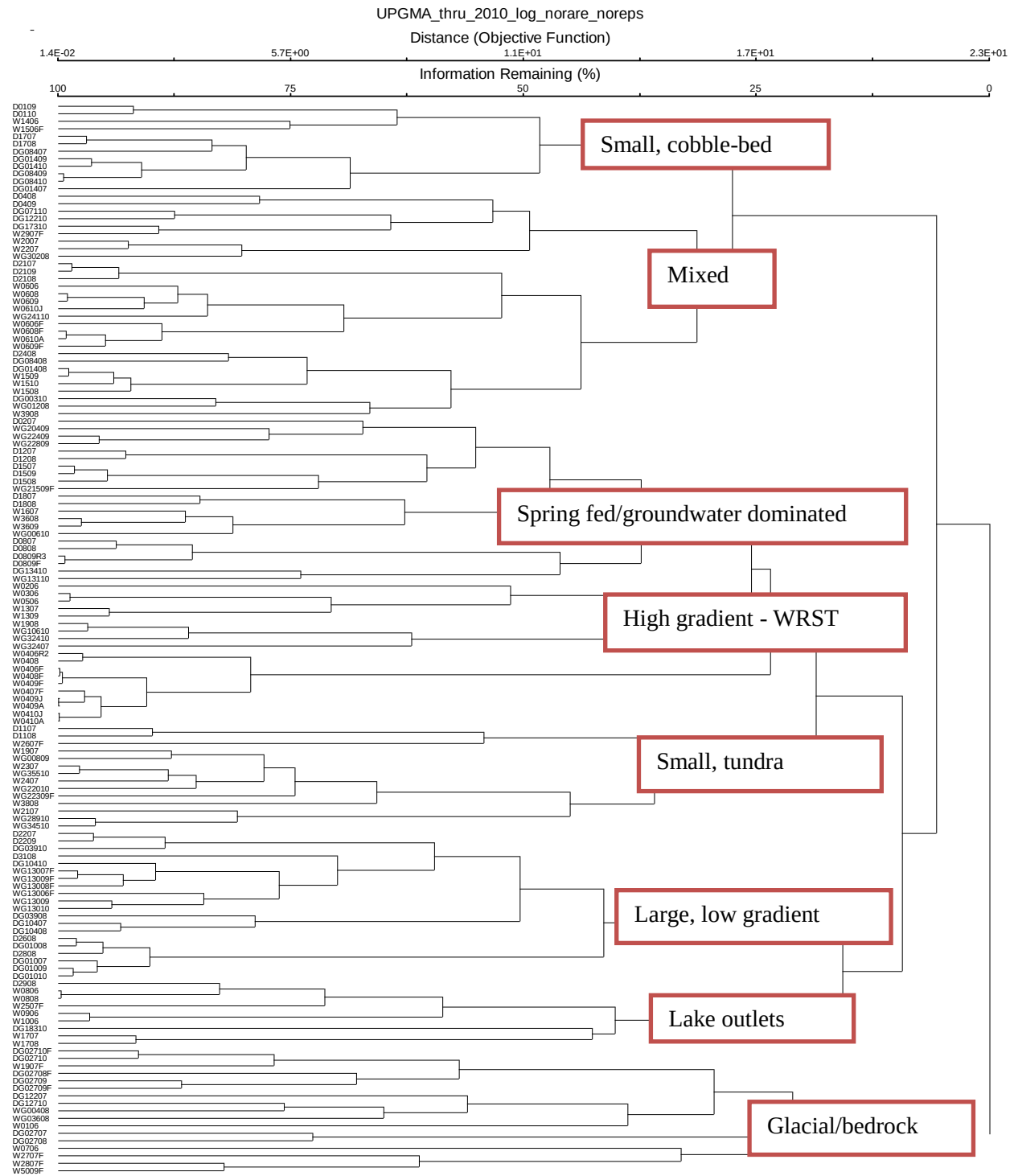


Figure 4. Flexible- β UPGMA cluster analysis ($\beta = -0.25$) of all invertebrate data from 2006-2010. Abundances were log transformed and rare taxa (occurring in < 5% of samples) were ignored. Bray-Curtis dissimilarity was used as the distance metric. The sampling year follows the sitecode (e.g., W0106), and an “F” is appended if it was a September/October sample of that site for that year (a fall sampling event).

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

The uppermost division in the dendrogram (the split on the far right of Figure 4 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/bedrock” in Figure 4) are either from glacial rivers or streams with intermittent flow or unstable beds. Other significant divisions are illustrated in Figure 4, and many seem to be related to stability in some fashion. For example, all of the lake outlet samples from all years cluster together. Lake outlets tend to be very stable in terms of flow, temperature and water chemistry. Similarly, all of the larger rivers are found on this side. Most of the spring-fed streams and streams that appear hydrologically stable based on the prevalence of bryophytes also cluster with these streams. Broadly speaking, these results are consistent with those of Milner *et al.* (2006), who found a substantial effect of stream stability on community composition and interannual variation in DENA.

Because the particular clustering algorithm used can affect the outcome of the cluster analysis, the same data were used in a second cluster analysis, this time employing the “group-average” clustering method. Like flexible- β , group-average clustering tends to be space-conserving (*i.e.*, the dendrogram preserves the pattern among sample units in the original data matrix) and is compatible with Bray-Curtis distance (Peck 2010). The dendrogram resulting from group-average clustering was similar in structure to the dendrogram depicted in Figure 4 (not shown).

NMDS is a free ordination technique that is particularly suited to analysis of ecological data, which often fail to meet the assumptions of other multivariate analytical methods (Peck 2010). An NMDS ordination of the CAKN invertebrate data from 2006-2010 is shown in Figures 5 and 6. The best solution was 3-dimensional (stress was 17 and significantly less than a solution with randomized data); however, only the first two dimensions are depicted. The ordination has an r^2 of 0.78 which indicates that much of the variation in the original data matrix is captured accurately. As was the case for cluster analysis, no geographic pattern is apparent (compare the distribution of red [WRST] and green [DENA] symbols). This suggests that differences in community composition among these sites are largely driven by local-scale environmental filters (*sensu* Poff 1997). Ecologically similar sites tend to be grouped in the ordination. Two of these groups are depicted in Figure 5. Lake outlets (stable) and glacial rivers (unstable) are grouped on opposite ends of horizontal axis, suggesting that this axis is correlated with some measure of ecosystem stability. Note that although all of the samples from glacial and unstable streams are found together at the right end of 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. In addition to providing insight into ecological

patterns among sites (spatial variation), NMDS can also be used to assess temporal variation and trends in community composition at individual sites. In Figure 6, all of the samples collected between 2006 and 2009 at each of three sites (Rock Creek, Little Stony Creek and Chalk Creek) are circled. It can be seen that the variation between samples collected at any given site (temporal variation) is substantially less than variation between sites (spatial variation), consistent with what other analyses have indicated. Successional vector analysis was conducted for all sites with at least three years of data. This method can reveal whether year-to-year changes in composition are directional or locally random. Directional changes in ordination space may indicate an ongoing shift in composition, *i.e.*, a trend. None of the 12 sites analyzed showed any directional change. As more data are collected, these methods will become more useful in assessing both spatial and temporal patterns in the distribution of invertebrates.

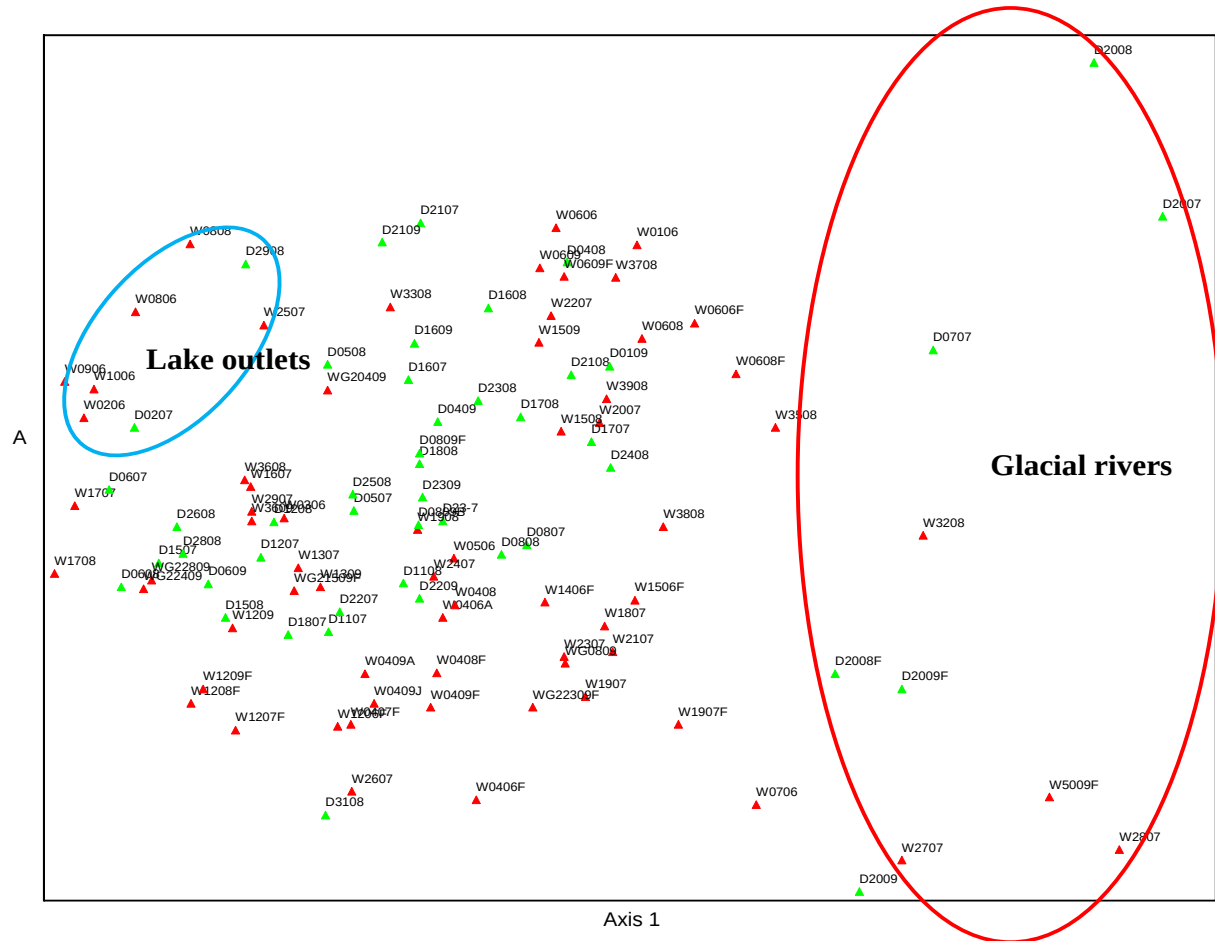


Figure 5. NMDS ordination of invertebrate data from 2006-2010. Abundance data were log transformed to reduce the effects of extreme values. Bray-Curtis dissimilarity was used as the distance measure. Site/year codes are the same as in Figure 4 and are listed in the appendix. DENA sites are shown in green, WRST sites in red. Axes 1 (horizontal) and 2 (vertical) are shown.

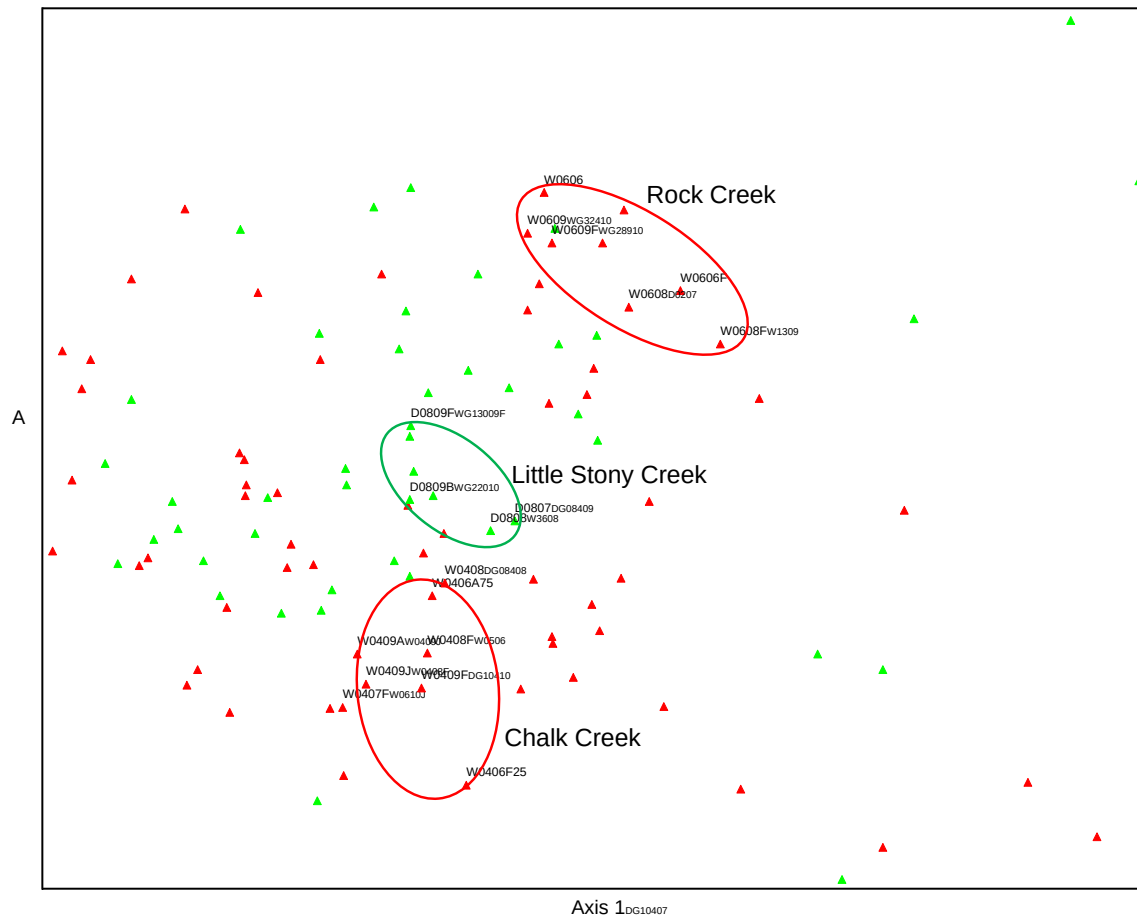


Figure 6. NMDS ordination of invertebrate data from 2006-2010 showing year-to-year variation at 3 sites.

Development of RIVPACS biological assessment models

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

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

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

In collaboration with Jeff Ostermiller, a bioassessment consultant, the CAKN has begun to explore the utility of these methods for detecting change in network stream ecosystems. A preliminary model, constructed using data collected from 2006-2009, 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. 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 (125 taxa) from 70 calibration sites in DENA and WRST, we recently began developing a RIVPACS model for the CAKN. We also built a null model (Van Sickle *et al.* 2005), which ignores environmental gradients, to test the improvement provided by the model. Finally, we built a modification of the RIVPACS model which uses an index of Bray-Curtis dissimilarity rather than taxa lists to quantify change in the macroinvertebrate assemblage (Van Sickle 2008). This modification has several potentially advantageous features, including an increased sensitivity to changes in composition that are not accompanied by changes in richness, and the

inclusion of rare taxa (which often decrease the sensitivity of traditional RIVPACS models and so are excluded).

Cluster analysis (flexible- β UPGMA) was used to separate the 70 sites into 6 groups based on biological similarity. Random forest discriminant analysis was applied to develop a discriminant model that could predict new sites into the biological groups based on environmental gradients, and used to calculate posterior probabilities of group membership for each calibration site. Ideally this procedure would be repeated with a set of validation sites, but at present we do not have sufficient data. Nevertheless, calculating the O/E scores for the calibration sites provides a measure of both precision and bias in the model. Table 13 lists the globally important predictor variables from the random forest discriminant analysis. There are 3 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 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.

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. 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

Figure 7 shows the distribution of O/E scores for the calibration data set, obtained by running the calibration site data through the finished model. The mean O/E score was 1.07, indicating a slight bias to the model in that on average it predicts more taxa than were actually observed. The precision of the model, as represented by the standard deviation of reference site scores, was 0.17, which is a substantial improvement over the pilot model (SD = 0.24) and is comparable to

other large-scale RIVPACS models. One factor affecting the precision of the model is the combination of very high heterogeneity and a relatively small number of sites. The dataset contains 125 unique taxa, many of which are found at only a few sites, and 2/3 of which never had a probability of capture (P_c) above 0.5 at any site. Taxa with a $P_c < 0.5$ (that is, relatively rare taxa) are normally removed from the model as a way of increasing model performance. This makes generating accurate predictions challenging. To increase the number of sites in the model, we have obtained additional data from both the EPA (46 sites) and the U.S. Geological Survey (21 sites). Although not all are in CAKN parks, inclusion of these sites, in addition to data from another 13 network sites sampled by CAKN in 2011, should provide additional improvements to the precision of the model.

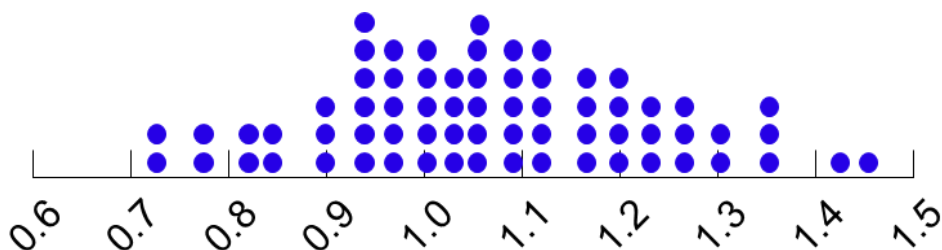


Figure 7. Distribution of O/E scores for calibration sites. Each dot represents a single site.

A second measure of model performance is model bias. Figure 8 shows a plot of modeled (E) vs observed (O) invertebrate taxa at each of the 70 calibration sites. Although the values are well correlated ($r^2 = 0.85$) and the distribution is not significantly different from the 1:1 line, it can be seen that O values are consistently above the 1:1 line for all E values greater than ~12. This indicates a slight positive model bias, particularly for more diverse streams. Once again, this is probably related to the need to incorporate data from additional sites to help mitigate the effects of the high biological heterogeneity in the dataset.

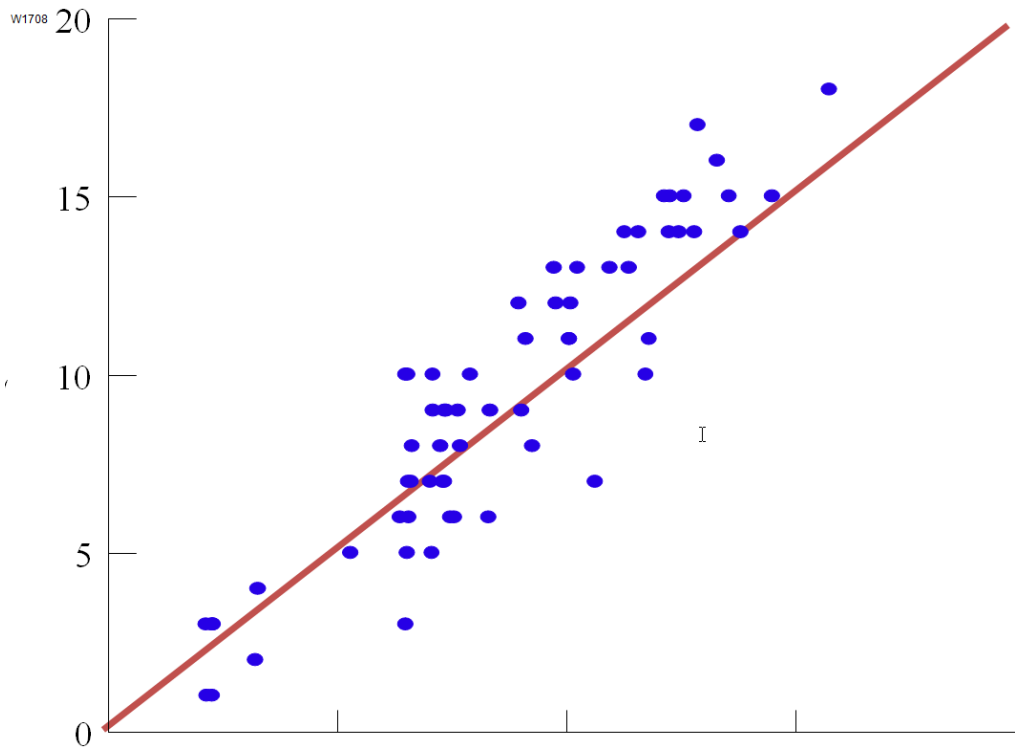


Figure 8. Plot of O vs. E for calibration sites. $R^2 = 0.85$, not significantly different from the 1:1 line.

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.

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