# Introduction

The Arctic Ocean represents 34% of the world’s coastline (Lantuit et al. 2012), yet is among the least studied marine ecosystems in the world due to inclement weather and inaccessibility. The shallow, broad coastal shelves in the Arctic comprise a much higher proportion of the coast compared to other oceans (Eakins and Sharman 2010), and are not accessible by deep-drafted, ocean-research vessels, causing nearshore ecotypes in Arctic regions to be under-studied. The nearshore regions of the Arctic support a robust fish community comprised of marine, freshwater, and diadromous species of all life stages that subsist in the seasonally estuarine conditions. There are approximately 211 circumpolar marine fish species in 39 families (Mecklenburg et al. 2011), as well as approximately 99 freshwater or diadromous fish species in 17 families (Reist et al. 2006) that inhabit Arctic waters, many of them found in the coastal Beaufort Sea along the northern Alaskan coast. Despite this diversity, there are few aquatic long-term studies in the Arctic (Kortsch et al. 2012), with most research focusing on either permafrost and resulting greenhouse gas emissions from thawing or high-trophic organisms (Fritz et al. 2017).

Climate models predict that the effects of climate change will be felt most acutely in the Arctic, with sea surface temperatures predicted to rise more than in temperate latitudes: mean annual temperatures are predicted to outpace global averages rising 3–10°C over the next 80 years (IPCC 2014). Expected changes are much broader than just temperature; for example, observed mean annual sea-ice spatial extent has been decreasing 3.5–4.1% per decade and precipitation is expected to increase 30–50% (IPCC 2014). A reduced duration of shore-fast sea ice coverage also means that winter storms will likely erode shores more quickly, with coastlines in many locations along the Beaufort Sea expected to increase in erosion rates 2 m/year and up to 25 m/year in many locations, while barrier islands may erode completely (Jones et al. 2009; Lantuit et al. 2012; Gibbs and Richmond 2015). The loss of sea ice coverage was predicted to increase benthic light budgets (Clark et al. 2013), increasing primary productivity in polar regions. However, the reduction in sea ice in the Beaufort Sea has increased suspended sediment load (caused by an increase in coastal erosion rates), decreasing benthic and water column primary production (Bonsell and Dunton 2018). Such changes in environmental conditions of Arctic nearshore regions holds implications for the ecological responses of local fish species.

Fishes are greatly affected by fluctuations in surrounding environmental conditions such as water temperature, salinity, and light availability. As poikilotherms, fish have internal temperatures that are regulated by their environment, which causes surrounding water temperatures to greatly affect their rate dynamic parameters (Pauly 1980). As a result, fish exhibit behaviors to seek environmental conditions that optimize growth and survival (Cushing 1990; Monaghan 2008). Such parameters may vary by life stage, especially for diadromous fishes (Werner and Gilliam 1984). Conditions outside of this thermal range can be lethal, but it is often difficult to determine sub-optimal or sub-lethal effects (Coutant 1987). Within estuarine areas, fluctuations in salinity are another important environmental variable, as osmoregulation can incur a significant energetic cost (Bœuf and Payan 2001). Both marine and diadromous fishes (e.g., gadids and salmonids, respectively) living outside of their optimal salinity limits can experience substantial reductions in growth or higher natural mortality (Arnesen et al. 1993; Dutil et al. 1997). Unique to the Arctic Ocean is the persistent presence of sea ice and its effect upon the local ecosystem. Many ice-associated taxa (e.g., calanoid copepods and amphipods) exist at the sea ice edges (Bradstreet and Cross 1982), supporting fishes such as Arctic Cod *Boreogadus saida* that prey upon such species (Gradinger and Bluhm 2004). The dynamic nature of the Arctic means that resources are highly variable and patchy at both spatial and temporal scales (Power 1997). These conditions cause several Arctic fishes to adopt migratory life histories to utilize multiple habitats, leading to a higher chance of encountering favorable conditions (Craig 1984; Roux et al. 2016). Thus, the intensity, duration, and variability of environmental factors plays a large role in determining fish species presence and abundance in a region, especially in the nearshore Arctic.

Nearshore areas are often important habitats that provide a wide diversity of trophic contributions, increased nursery production of juvenile fishes (Beck et al. 2001), and ontogenetic migration corridors (Sheaves et al. 2014). There are a variety of habitat types within nearshore habitats, which comprise a gradient of temperature and salinity conditions, helpful for fish species that prefer a specific niche. These environmental conditions can be reflected by the presence/absence and abundance of fish species because fish community composition can be a function of a temperature and other abiotic factors for both marine and freshwater species (Jackson et al. 2001; Collie et al. 2008). For example, the distribution of juveniles in estuarine conditions can be a function of turbidity (Blaber and Blaber 1980). The variability of an ecological community assemblage structure is often thought to be indicative of increased stressors acting upon the populations (Warwick and Clarke 1993). Resilience of ecological communities to persist during changes to their habitat or environment is a characteristic of healthy communities with high biological diversity and wide response diversity of these species (Peterson et al. 1998; Elmqvist et al. 2003).

Arctic whitefishes species like Broad Whitefish *Coregonus nasus*, Arctic Cisco *Coregonus autumnalis*, Least Cisco *Coregonus sardinella*, and Humpback Whitefish *Coregonus pidschian* are amphidromous and tolerant of moderate levels of salinity (Bond and Erickson 1985; de March 1989; Fechhelm et al. 1993). Due to the dynamic nature of resources and conditions in the Arctic, some Arctic fishes adopt migratory life histories to increase chances of encountering habitats with favorable conditions (Craig 1984; Power 1997; Roux et al. 2016). For example, Arctic Cisco found in Alaskan waters are hatched in the Mackenzie River, Northwest Territories, Canada and are transported east as juveniles via easterly winds pushing surface currents where they spend six to eight years rearing in Alaskan estuaries and rivers before returning westward to spawn in their natal waters within the Mackenzie River (von Biela et al. 2013; Zimmerman et al. 2013). Conversely, Least Cisco, Broad Whitefish, and Humpback Whitefish spawning populations are found in many of the rivers across northern Alaska (Craig 1984, 1989; Fechhelm et al. 1994). The juveniles of all four of these whitefish species spend summer months feeding in the estuaries and deltas found along the coastal Beaufort Sea and overwinter in deep-water pools or areas of upwelling in local rivers, especially the Colville River (Craig et al. 1985; Fechhelm et al. 1999; Seigle and Gottschalk 2013). Gadids, predominantly Arctic Cod, support the Arctic marine ecosystem with their prevalence and high energetic content (Gradinger and Bluhm 2004; Elliott and Gaston 2008; Harter et al. 2013; Majewski et al. 2016; Thorsteinson and Love 2016). Arctic Cod play a key linkage between abundant zooplankton such as calanoid copepods and amphipods and higher trophic organisms such as Black Guillemot *Cepphus grille*, ringed seals *Pusa hispida*, and beluga whales *Delphinapterus leucas* (Bradstreet and Cross 1982; Harter et al. 2013; Thorsteinson and Love 2016).

We assessed the effects of environmental variables upon the various fish species present within a nearshore semi-estuarine ecosystem of the Alaskan Arctic to quantify the relationship of these variables upon the abundance of fishes. We hypothesized that ongoing shifts in Arctic environmental conditions would bring about measurable changes in fish populations during 2001–2018. To quantify changes in fish community metrics we modeled 1) total species richness across and within sampling years, 2) binomial presence / absence of rare species over time, 3) effects of environmental variables upon species assemblage structure ordinations, 4) permutational multivariate analysis of variance (PERMANOVA) of environmental effects upon species assemblage structure, and 5) testing whether species assemblage structure has changed over time.

# Methods

The Beaufort Sea is a peripheral sea to the Arctic Ocean along the northern coast of Alaska and Canada. Coastlines along the Alaskan Beaufort Sea are typically salt marshes and slumping tundra, and coastal erosion rates appear to be increasing at many locations (Gibbs and Richmond 2015). Much of the coastal Beaufort Sea is shallow; for example, the 2-m isobath is 1 to 8 km from shore (Britch et al. 1983; Ross 1988). Barrier islands occur along much of the Alaskan Beaufort Sea coast, differentiating outside marine waters from the coastal waters. There are several large sources of freshwater inputs into the Beaufort Sea, including the Colville River (9.0 km3/year; USGS 2018a), the Sagavanirktok River (1.5 km3/year; USGS 2018b), and the Mackenzie River (325 km3/year; Yang et al. 2015). Prudhoe Bay is a semi-estuarine bay of the Beaufort Sea formed near the mouth of the Sagavanirktok River delta. The immediate surrounding coastal waters are shallow, with the 6-m water depth contour less than 5 km from most parts of natural, unaltered shore and several barrier islands are within 15 km of shore (Ross 1988). Much of the terrestrial environment around Prudhoe Bay has developed infrastructure for the extraction and processing of oil, with many permanent structures inland from the coast. In addition, several oil extraction and processing facilities have been constructed on man-made islands that are connected to shore with gravel causeways and bridge breaches (Ross 1988). The majority of the shoreline remains as natural tundra banks, although the rate of erosion has increased at many locations (Gibbs and Richmond 2015).

Beginning in 1981, daily fish monitoring has occurred annually along the coast near Prudhoe Bay, Alaska for approximately 8–10 weeks each summer during July and August, with the exception of the 1999 and 2000 field seasons. Since 2001, a standardized sampling methodology for collecting daily abundance and length data (for select species of subsistence importance) have been recorded in an annual database. Four fixed stations collect fish using double-ended fyke nets, located roughly east-west, spaced approximately 27 km apart (Figure 1). From west to east, these sites are identified as Site 220 (approximately 1 km west from the base of the West Dock causeway; sampled 2001–2018), Site 218 (on the west side of Prudhoe Bay at the West Beach drilling pad; sampled 2001–2018), Site 214 (at the Niakuk drilling pad on the tip of Heald Point; sampled 2002–2018), and Site 230 (located midway on the eastern side of the Endicott Causeway; sampled 2001–2018). In 2001, Site 231 was fished on the western side of the Endicott causeway to follow historical sampling locations, but due to changing bathymetry the site was replaced the following year with the current sampling location of Site 214. Site 231 was removed from all analyses.

At each of the four sampling locations, two fyke nets with an opening of 1.8 m by 1.7 m were set side-by-side, opening towards the coastline, with a 60-m blocker net leading to shore. A 15-m blocker wing was attached to the outer edge of each cod end. Using this bi-directional sampling method, the fyke nets could intercept and catch fish moving along the shoreline in either direction. All blocker lead nets and wings were constructed from 2.5-cm stretch mesh, while the fyke net mesh consisted of 1.27-cm stretch mesh. Three consecutive throats were located behind each 1.7-m frame opening, with the outermost throat having a functional width of 11.4 cm. Net specifications were consistently used throughout the duration of the study, with the exception of a modification in 2009 to add a vertical metal bar to the fyke net funnel to prevent incidental seal catches. Sampling sites were operated from approximately July 1 through September 1 each year, with the precise dates of installation and removal for each site varying each year. The latest date of first sampling was July 6 (2018) and the earliest date of last sampling was August 25 (2012).

Each net was checked daily and all fish were identified to species and enumerated. Occasionally, unsafe conditions prevented sampling at certain sites. After species identification, enumeration, and measurements of a subsample of select species, fish were released away and offshore from the cod-end openings to minimize recapture. All fish were identified using Mecklenburg et al. (2002), George et al. (2009), and Thorsteinson and Love (2016). Field sampling protocols were essentially unchanged from 2001 to 2018, except for the addition in 2017 of length measurements from new fish species. Abundance data were recorded by net site and side for each species; for all analyses, both net sides were aggregated. Fish abundance data were pooled then scaled by combined sampling effort to get biweekly and annual catch per unit effort (CPUE) for each species. Two fish that were unable to be identified to species and 24 hybridized ciscoes were excluded from all analyses.

Environmental variables examined were water temperature (°C) and salinity (ppt) from midway through the water column, wind directionality and speed (east/west; km/h), annual sea ice extent (maximum overwinter area; km2) and duration (number of days from freeze up to melt date), discharge for the Sagavanirktok River (recorded as ft3/s, converted to m3/s). Results of a correlation matrix showed no variables were highly correlated (>0.5) with each other on either biweekly or annual scales.

Salinity (ppt) and water temperature (°C) data were collected daily during each sampling event at each site near the fyke net cod ends using a calibrated handheld YSI 30 salinity / conductivity / temperature meter (YSI Inc., Yellow Springs, Ohio) at the bottom, mid-water column, and just below the surface.

Meteorological data was collected hourly by the National Weather Service (NWS) at the nearby Deadhorse Airport data collecting site (PASC), located 12 km inland from Prudhoe Bay. Wind speed was measured in km/h; wind directionality was recorded as cardinal direction (0–360) and averaged using the circular mean, then converted from polar to cartesian coordinates to get the East-West unit vectors, using the formula:

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Sea ice duration and extent are from the Sea Ice Index, Version 3 from the National Snow and Ice Data Center (NSIDC) in Boulder, Colorado (Fetterer et al. 2018). Sea Ice Index data are daily summaries of remote sensing satellite data derived from two passive-microwave bands: the Near-Real-Time DMSP SSMIS Daily Polar Gridded Sea Ice Concentrations and the Sea Ice Concentrations from Nimbus-7 SMMR and DMSP SSM/I-SSMIS Passive Microwave Data. Daily sea ice coverage maps from the Sea Ice Index show the approximate extent across the entire polar region. Annual dates of sea-ice break up and formation will be estimated using the first breakup dates and final freeze-up for the central Beaufort Sea coast. Determining the precise dates will be done visually using the daily NSIDC GeoTIFF of sea ice coverage, inspected in the open-source software QGIS, version 3.0.1 (QGIS Open Source Geospatial Foundation Project 2018).

Measurements of daily Sagavanirktok River discharge come from a United States Geological Survey (USGS) gaging station upriver from the river mouth, near Pump Station 3, recorded in ft3/s and converted to m3/s.

All data were imported and analyzed using the statistical program R (R Foundation, https://www.R-project.org, Version 3.5.2). To investigate trends in annual and seasonal species richness, we used Generalized Additive Models (GAMs) to account for potential curvature in the models as GAMs are typically responsive to non-linear response curves commonly found in ecological response scenarios (Guisan et al. 2002), using package “mgcv” in R.

Total number of species were defined as all sites combined, in a given period of time (each two-month season was split into four biweekly periods). Top fitting model fits were determined using Akaike’s Information Criterion (AIC) and the “dredge” function from the R package “MuMIn”. Smoothing functions were restricted to k=3 to prevent overfitting of the model and were also tested against null models and models without any smoothing terms. is the biweekly sampling period that was assigned as four events evenly distributed during the sampling season (i.e., period 1 = July 15 and before, period 2 = July 16–31, period 3 = August 1–15, and period 4 = August 16 and afterwards).

To determine whether there were changes in catch rates of 15 “rare” species (defined as species ≤ 100 individuals caught in all years combined) was assessed using binomial Generalized Linear Models (GLMs) of presence / absence for each rare species:

where is the mean value of species *j*, and is the effect for each Station *s*. is the biweekly sampling period (1–4).

Multivariate analyses were conducted using the “vegan” package in R (Oksanen et al. 2018). This package allows for the multivariate analysis of quantitative environmental variables upon the responding species assemblages. To prevent rare species from dominating multivariate analyses (CITE XXXXX), we restricted analysis to the 17 species whose abundance was greater than 100 fish, all years combined. All multivariate analyses were performed on aggregated catches on both an annual and biweekly scale (i.e., biweek number 1 includes all sampling on July 15 and before; biweek number 4 includes all sampling on August 16 and after). Response variables for multivariate analyses were species catch per unit effort (CPUE) that were fourth-root transformed then standardized to percent of the maximum transformed catch. A dissimilarity matrix of the multivariate catch data was created using Bray-Curtis distances. The Bray-Curtis dissimilarity matrix was then reduced to three dimensions using non-metric multidimensional scaling (nMDS) using k=3.

To determine which environmental variables were the optimal subset to explain the observed catches, we used a Mantel test (vegan command: bioenv()) on the Bray-Curtis distances with a Euclidean distance metric and Spearman correlation method. Significance testing of environmental variables was performed using vector and factor fitting upon MDS1 & MDS2 (vegan command: envfit()).

The most parsimonious model of environmental variables upon the species CPUE dissimilarity matrix was determined by Permutational Analysis of Variance (PERMANOVA) using environmental variables (for each variable and all combined variables). PERMANOVA models were run for both annual and biweekly aggregations; biweekly periods that had gaps in data coverage in environmental monitoring were excluded from analysis. Environmental variables were square root transformed for PERMANOVA modeling. The top fitting PERMANOVA model was determined using backward stepwise model selection, with improved model fit defined as a reduction in residual R2, using marginal results (vegan command adonis2()).

The full model for PERMANOVA was:

where are the Bray-Curtis dissimilarity matrix of transformed and standardized CPUE for all 17 species, is the East-West wind vector (-1 = 270° West wind; 1 = 90° East wind).

Similarity percentile (simper()).

Before modeling, all environmental variables will be assessed in a correlation matrix. Variables that were highly correlated (>0.6) were removed to prevent multicollinearity. Removed wind dir and

Modeling of the MDS time series data compared linear models, nested linear effects models (station effects by year), Generalized Additive Models (GAM), and tested for optimal breakpoints in the time series using Chow Test for structural breaks (CITE XXXXXXX).

These GLS models will be used when the response is not necessarily expected to be non-linear (e.g., modeling the effect of time upon species richness). The autoregressive models will test for normality, and then assume any errors to be Gaussian.

All significance testing was be done at the α = 0.05 level.

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