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Population decline of northern dusky salamanders at Acadia National Park, Maine, USA

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A B S T R A C T

We investigated and reviewed the current and historic distribution of northern dusky salamanders (*Desmognathus fuscus fuscus*) in Acadia National Park (ANP), Maine, USA during 1938–2003. Historical data indicate that northern dusky salamanders were once widespread and common in ANP. We conducted intensive surveys for stream salamanders during 2000–2003 and observed only two adult northern dusky salamanders on one stream. No eggs or larvae were observed. Although the cause of the observed population decline is unknown, we identify multiple potential stressors including stocking of predatory fishes, fungal pathogens, substrate embeddedness, and widespread pollution (i.e., from atmospheric pollutants) of surface waters at ANP. Our data suggest that ANP streams may no longer be suitable for northern dusky salamanders. This investigation is the first to document the decline of a stream dwelling amphibian species in a national park (i.e., areas that are not subject to obvious habitat loss or major changes in land use) with widespread mercury contamination of its surface waters.

1. Introduction

The physical conditions of habitats are critically important in lotic environments. In general, headwater streams are considered more susceptible to watershed degradation than other ecosystems (Power et al., 1988) primarily because of their small size and close linkage with both atmospheric and local watershed conditions. In aquatic ecosystems, signs of environmental stress and deterioration are often initially detected at the population level and usually affect sensitive species first (Odum, 1992; Rapport and Reiger, 1995).

Worldwide amphibian declines are well documented (Pounds et al., 1997; Houlahan et al., 2000; Stuart et al., 2004; Beebee and Griffiths, 2005). These declines include populations from national parks and natural wilderness areas that are not subject to obvious habitat loss or major changes in land use (Fellers and Drost, 1993). Population declines have been attributed to multiple stressors (Semlitsch, 2003) including habitat loss and degradation, UV–B radiation, contaminants including mercury (Hg), exotic species, global climate change, and fungal pathogens and disease (Blaustein et al., 1994; Stebbins and Cohen, 1995; Alford and Richards, 1999; Semlitsch,

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2003; Unrine et al., 2004; Stuart et al., 2004; Bank et al., 2005, in press; Beebee and Griffiths, 2005). Amphibians play an important role in many aquatic and terrestrial environments, and therefore their declines may have significant, negative effects on ecosystem functions (Burton and Likens, 1975; Gibbons, 1988; Stebbins and Cohen, 1995; Davic and Welsh, 2004) such as nutrient cycling in aquatic and terrestrial ecosystems.

Anthropogenic habitat loss and degradation (i.e., from contaminants, pollution and other sources) are believed to be primary threats to the health and persistence of amphibian populations (Alford and Richards, 1999; Semlitsch, 2003; Stuart et al., 2004; Beebee and Griffiths, 2005). Amphibians in general are considered to be sensitive to disturbance in both aquatic and terrestrial environments, primarily because of their complex life histories, specialized physical adaptations, and micro-habitat requirements (Bury, 1988; Vitt et al., 1990; Wake, 1990; Olson, 1992; Blaustein, 1994; Blaustein et al., 1994; Stebbins and Cohen, 1995; Davic and Welsh, 2004). In contrast to vernal pool amphibian species, whose larval period is short, stream salamander larvae are often slow-growing (Petranka, 1998). Larval development of these salamanders occurs in streams, and larvae are strictly aquatic during this life history stage, relying on gills for breathing. However, after metamorphosis, adults can often be found in streamside habitats in leaf litter or at the interface of the stream channel under partially submerged rocks. Their long life span, high relative abundance and vagility, philopatric behavior (Daugherty and Sheldon, 1982; Welsh and Lind, 1992), and stable populations (Hairston, 1987) make stream salamanders ideal indicators of environmental stressors including contaminants in lotic ecosystems (Welsh and Ollivier, 1998; Bank, 2003; Southerland et al., 2004; Bank et al., 2005, in press).

The northern dusky salamander is a widespread and ubiquitous species found throughout eastern North America and has been observed throughout most of mainland Maine, except for the northeastern-most corner (Hunter et al., 1999). The larval period of the dusky salamander is approximately 9–14 months and is influenced by local abiotic and biotic conditions including climate, temperature, food availability, competition and predation (Petranka, 1998). The specific objective of this study was to synthesize existing data related to the spatial and temporal distribution patterns of northern dusky salamanders at Acadia National Park (ANP), Maine. We also discuss the potential causes of population decline at ANP.

2. Methods

2.1. *Study site*

ANP (44°21′N, 68°13′W) encompasses 15,233 ha, with 12,260 ha on Mount Desert Island (MDI) and 2973 ha in surrounding parcels (Fig. 1). ANP has 26 mountains, and ~20% of the park is classified as wetland habitat including marshes, lakes, ponds, streams (elevation range 50–250 m), vernal pools, swamps, and bogs (Calhoun et al., 1994). The park also contains salt marshes, marine aquatic beds, and intertidal shellfish flats. Terrestrial habitats include peatlands, coniferous forest, and upland and riparian deciduous forests. The area is dominated by white spruce (*Picea glauca*), red spruce (*Picea rubens*), and balsam fir (*Abies balsamea*). Dominant deciduous tree species include paper birch (*Betula papyrifera*), trembling and big-toothed aspen (*Populus tremuloides* and *Populus grandidentata*), red maple (*Acer rubrum*), and red oak (*Quercus rubra*). In 1947, a human-caused fire swept through Bar Harbor, Maine, severely burning 6880 ha of northeast MDI. In ANP, aquatic salamanders (*Eurycea bislineata bislineata*, *Desmognathus fuscus fuscus*) inhabit small stream and seep environments, although *D. fuscus fuscus* are now considered extremely rare while *E. bislineata bislineata* is still common and widely distributed throughout ANP (Bank, 2005).

2.2. *Fish stocking in Acadia National Park*

The objectives of the National Park Service's (NPS) recreational fisheries program are to protect native fish species and aquatic ecosystems and provide recreational angling opportunities for the public. In the nutrient limited lakes of Acadia NP, this requires a careful balance between protection of native species, and the stocking of fish to enable viable recreational fishing by the public. The NPS together with the Maine Department of Inland Fisheries and Wildlife (MDIFW) regulate and manage freshwater fishing in Acadia National Park, although the MDIFW is largely responsible for supervising the fish stocking programs. The fish communities of Acadia have been altered by humans for >100 years. Increases in recreational fishing during the second half of the 1900s led to an extensive fish introduction program, including the stocking of native and non-native predatory fishes (Stone et al., 2001). Recently stocking programs have focused on the introduction of salmonid species (Moring, 1999), and only Bear Brook Pond and Duck Pond in Acadia have never been stocked (Lê and Moring, 1999; Stone et al., 2001).

2.3. *Atmospheric pollution at Acadia National Park*

Sulfur dioxide, Hg, and nitrogen oxide from power plants and other sources are likely having adverse effects on surface waters in ANP (Bank et al., 2005). Acadia's thin rocky soils provide little buffering capacity against acid rain damage to headwater streams, ponds, and lakes and Hg contamination of ANP biota is well documented (Burgess, 1997; Bank et al., 2005, in press). Acidification of surface waters, including headwater streams, can reduce species diversity and can exert adverse effects on aquatic biota such as fish and salamanders, primarily as a result of direct effects of decreases in pH that can cause elevated concentrations of highly toxic aluminum and Hg (Schindler et al., 1985; Cronan and Schofield, 1990; Driscoll and Postek, 1995; MacAvoy and Bulger, 1995; Driscoll et al., 2003; Bank et al., 2005). Episodic acidification of headwater streams, likely caused by the interaction between marine salt deposition and ANP's acidic soils, has been documented with a pH < 5.0 (Kahl et al., 1985; Heath et al., 1992). The pH of precipitation in the park averaged 4.5 (range 4.4–4.6) during 1982–1990 (Kahl et al., 1993) and fog at ANP often has a pH of 3.0 (Kahl et al., 2002; NPS, unpublished data). In general, despite reductions in sulfur dioxide emissions and sulfate deposition since the 1990 Clean Air Act Amendment, the pH and acid neutralizing capacity of ANP's surface waters remains relatively unchanged (Kahl et al., 2003).

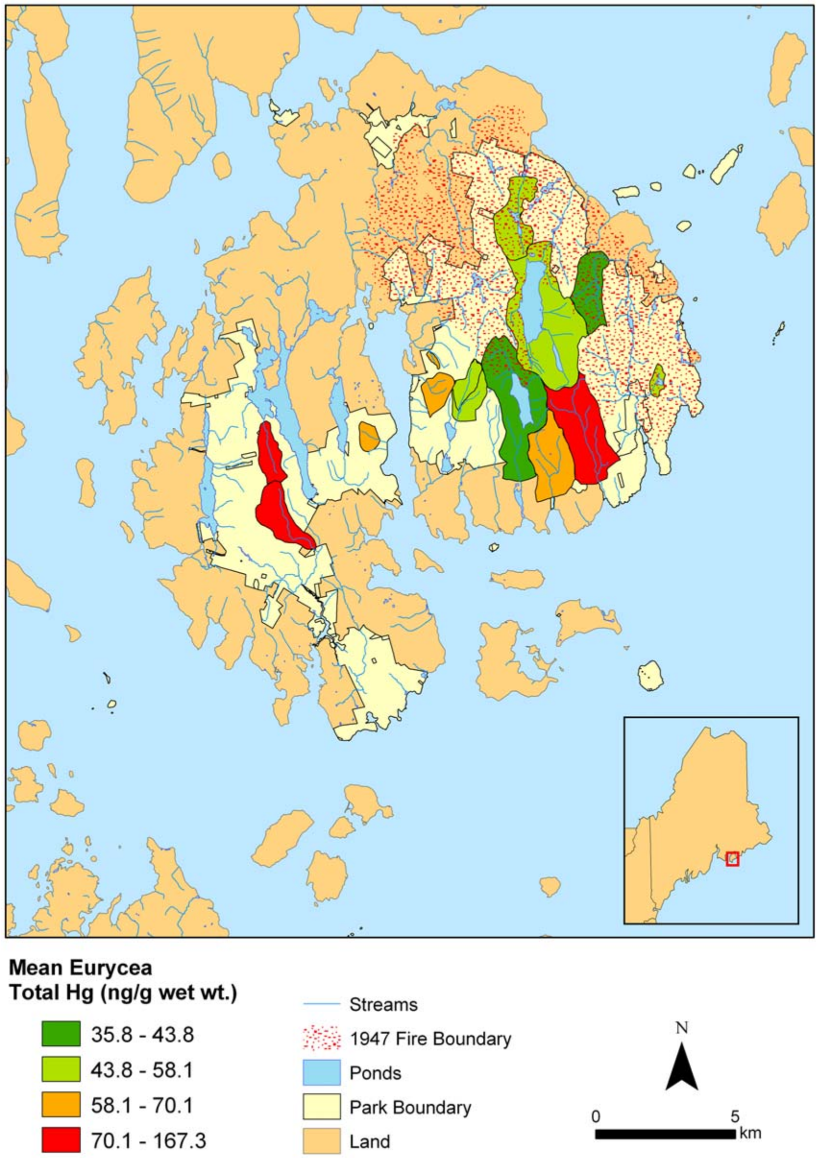

Fig. 1 – Spatial distribution of the average total Hg (ng/g wet wt.) concentrations in *Eurycea bislineata bislineata* larvae from Acadia National Park, Maine, USA, June–July, 2001–2002.

Atmospheric deposition is believed to be the primary source of Hg in the northern US, with deposition rates ranging 5–10 $\mu g/m^2$ yr (Fitzgerald et al., 1991). Wet deposition of Hg at ANP has averaged 7.9 $\mu g/m^2$ yr since 1995 (Hg Deposition Network (MDN)/National Atmospheric Deposition Program (NADP), 2004). Rates of Hg accumulation to the sediment at two locations in ANP were 100–200 $\mu g/m^2$ yr in the 1980s, suggesting that a large amount of dry Hg input is not measured by the wet-only MDN collector (Norton et al., 1997). These Hg accumulation rates are comparable to those reported from urban lakes (Engstrom and Swain, 1997) and presumably reflect deposition of Hg from upwind sources including metropolitan regions and solid waste incinerators in the northeastern United States. These high deposition rates at ANP are a significant concern for the NPS (Maniero and Breen, 2004), especially considering the Class I air quality status (i.e., requiring the highest level of protection under the Clean Air Act) of the Park, and the public and environmental health implications of Hg pollution.

2.4. *Assessment of historical data and field sampling*

Information on the historic distribution of northern dusky salamanders in ANP was gathered from published literature sources (Manville, 1938, 1939; Davis, 1958; Favour, 1963; Coman, 1987). We used field sampling methods described by Lowe and Bolger (2002) for stream salamander surveys during May–September of 2000–2003. We surveyed streams ($n = 37$ of 41, or 90%, of the named streams in ANP; Table 1) to document the presence or absence of northern dusky salamanders with 2–3 visits per stream per year. Stream surveys involved turning over rocks, woody debris, and other cover objects in all mesohabitat types (riffle, run, pool) while moving upstream to minimize disturbance. Streamside habitats were

Table 1 – Sampling site location and habitat characteristics of streams searched for northern dusky salamanders during May–September, 2000–2003, in Acadia National Park, Maine, USA

Stream name	UTM-N	UTM-E	Dissolved oxygen (mg/l)	Specific conductance (μs/cm)	pH	Temperature (°C)	Mean (±SE) channel width (cm)
Sargeant Brook	0556530	4911045	7.32	19.6	5.89	14.1	284.7 (±27.9)
Sargeant Drive Brook	0555748	4909783	10.32	154.7	6.43	14.0	333.0 (±17.3)
Bowl Brook	0564069	4909387	3.67	42.0	6.02	17.5	237.5 (±20.6)
Man-o-War Brook	0554294	4907537	8.38	32.2	6.97	15.3	201.3 (±30.5)
East Schoodic Brook	0573756	4911952	7.75	69.3	4.58	15.6	105.2 (±18.6)
West Schoodic Brook	0573757	4912044	7.70	70.3	4.49	16.0	164.7 (±23.2)
North Norumbega Brook	0555424	4908373	11.57	38.4	5.27	12.6	84.0 (±24.9)
Norumbega Brook	0555397	4908337	7.46	42.1	4.95	12.7	98.5 (±18.6)
Deer Brook	0559009	4909946	5.59	35.8	6.03	11.0	241.5 (±28.9)
North Jordan Brook	0559045	4909952	8.46	34.4	6.68	17.9	332.5 (±30.1)
Northeast Jordan Brook	0559557	4909468	9.56	50.1	6.75	12.7	253.8 (±56.2)
Great Brook	0550107	4908143	10.56	26.0	6.57	13.3	302.6 (±11.1)
Southwest Pemetic Brook	0560014	4907921	7.65	20.0	5.36	10.0	91.2 (±22.9)
Duck Brook	0560300	4914551	8.65	8.1	6.58	17.9	298.7 (±32.5)
Lower Cannon Brook	0563099	4909170	6.59	31.7	5.98	14.0	201.5 (±15.8)
Cromwell Brook	0563203	4913297	8.33	83.4	6.62	18.0	204.8 (±19.2)
Little Harbor Brook	0558417	4906742	10.73	59.8	7.30	16.2	345.5 (±68.1)
Jordan Brook	0559284	4906117	8.06	40.2	6.29	17.4	434.3 (±49.4)
Steward Brook	0548323	4906221	9.51	38.4	5.59	13.3	212.8 (±29.3)
North Steward Brook	0548323	4906603	10.17	33.4	5.95	13.3	154.0 (±19.2)
Duck Pond Brook	0549537	4908823	9.80	31.6	5.68	13.8	156.2 (±37.5)
Lurvey Brook	0551341	4902788	9.35	49.0	4.98	14.1	128.5 (±17.7)
Bubble Brook	0560532	4910973	8.13	33.5	6.42	17.1	226.7 (±22.4)
Breakneck Brook	0559483	4916963	7.90	21.6	6.50	15.8	380.7 (±48.7)
Kebo Brook	0562059	4914199	10.87	31.1	6.94	13.3	447.2 (±36.9)
Razorback Brook	0551375	4904036	9.92	27.3	6.06	14.8	237.3 (±37.1)
Lower Hadlock Brook	0556838	4907626	10.09	42.5	6.66	12.6	382.0 (±41.1)
Hadlock Brook (East)	0557237	4908892	9.77	39.8	6.68	14.2	272.8 (±44.5)
Hadlock Brook (West)	0557168	4908892	9.89	44.5	6.94	14.4	212.3 (±25.9)
Otter Creek	0563515	4908268	9.64	54.4	7.02	18.4	611.3 (±74.1)
Hunter's Brook	0562093	4906152	11.0	49.1	7.16	15.4	466.7 (±16.5)
Heath Brook	0551133	4902684	8.88	41.3	5.47	21.3	237.5 (±24.3)
Stanley Brook	0560428	4905632	11.67	81.3	7.30	14.0	395.8 (±22.4)
Upper Marshall Brook	0551618	4904522	7.26	54.0	5.18	12.6	78.2 (±12.3)
Richardson Brook	0557100	4912607	–	–	–	–	–
Hodgdon Brook	0549045	4908024	–	–	–	–	–
Chasm Brook	0558072	4912052	–	–	–	–	–

searched in a similar fashion, extending out to 5–10 m into the adjacent riparian zone. The length of stream survey reaches varied, although all were ⩾500 m. In areas with reduced visibility we used Aquascopes (Seal Cove, ME) to search for larval northern dusky salamanders. Crocker (2003) and Bank (unpublished data) reported little observer bias for two individual sets of observers sampling stream salamander larvae in ANP and Shenandoah National Park.

2.5. *Spatial juxtaposition of mercury in Acadia watersheds*

To map the spatial distribution of Hg concentrations in lotic biota throughout ANP we used ArcGIS 9.0 (ESRI Inc., 2004) to delineate watersheds with existing *E. bislineata bislineata* larvae Hg data (Bank et al., 2005). Watersheds were calculated using the lowest point in the watershed where a larval salamander was captured as the "pour point" for delineating the watershed and its area. We then used the watershed tool in ArcGIS 9.0 to delineate the watershed boundaries. We intersected the catchment boundary with the average Hg concentration level of larval individuals inhabiting the stream for the selected watershed using the join feature in ArcGIS 9.0.

For accuracy we compared our results to watersheds delineated by Perrin (1997). In three cases the watershed boundaries could not be delineated by the ArcGIS watershed tool due to their small size and the limited resolution of the 10 m digital elevation data. For these watersheds we relied on drainage characteristics determined from both the hillshade feature in ArcGIS 9.0 and USGS topographical maps to create the watershed boundary.

3. Results and discussion

3.1. *Historical assessment of population range*

Historically, the northern dusky salamander was reported by Manville (1938, 1939) from six localities primarily on the east side of MDI, and Davis (1958) recorded 16 individuals from six streams, five of which were on the east side of MDI. Favour (1963) reported them at six sites on the east side and one site on the west side of MDI, and Coman (1987) reported northern dusky salamanders in stagnant pools and ditches in Acadia. During a comprehensive amphibian survey of Acadia stream habitats in the mid-1950s, all age classes of northern dusky salamanders were commonly found in streams with cobble substrates and adults and larvae were widely distributed throughout ANP (Davis, 1958).

During the 2000–2003 surveys, investigators searched the preferred habitat of the dusky salamander including in-stream habitats and micro-environments along the edges of forested streams, springs and seeps (Petranka, 1998; Pfingsten and Downs, 1989). Duck Pond Brook, Cromwell Brook and Bubble Brook at ANP drain lakes or ponds and historically supported populations of northern dusky salamanders (Davis, 1958; Coman, 1987), however during extensive surveys from 2000 to 2003 we did not observe any dusky salamanders at these historically occupied sites. Only two adult northern dusky salamanders were observed (date of detection – May 28, 2002) on the lower reaches of Breakneck Brook during the entire survey period from 2000 to 2003 (Fig. 2). No dusky salamander eggs or larvae were ever observed during 2000–2003. The adult dusky salamanders were located near the stream edge on a reach with abundant cobble ranging in size from 3 to 30 cm.

It is likely that the sampling effort during 2000–2003 was much greater than the sampling effort of the investigators who collected the historical data which strengthens our conclusion that northern dusky salamanders have experienced a dramatic population decline at ANP (i.e., reports of a historically wide distribution of northern dusky salamanders with a lower sampling effort in contrast to an extremely low observation rate of this species with an intensive sampling effort). Additionally, we feel our survey data extends long enough (four spring–summer seasons during 2000–2003) to make inferences about the distributional change (Skelly et al., 2003) of the local population of dusky salamanders at ANP.

Historical accounts provide no quantitative data of northern dusky salamander biomass at ANP. However, the historic distribution data, and observations by Davis (1958) indicate that both northern dusky salamander adults and larvae were fairly widespread and common throughout ANP and were often observed on streams with abundant cobble substrates.

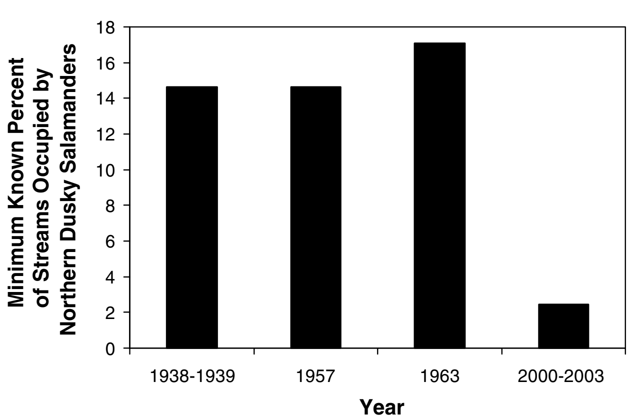

Fig. 2 – Minimum known percent of streams ($n = 37$) occupied by northern dusky salamanders in Acadia National Park, Maine, USA, 1938–2003. Historic data are from Manville (1938, 1939), Davis (1958), and Favour (1963).

While the northern two-lined salamander is still moderately common (Bank, 2003; Bank et al., 2005), the northern dusky salamander appears to be nearly extirpated from ANP. Reasons for its disappearance are unknown. Within its natural range outside ANP, the northern dusky salamander is still considered common and widespread (Hunter et al., 1999). Klemens (1993) and Orser and Shure (1972) both note that northern dusky salamanders have declined or been extirpated in many urbanized areas, where extensive paving has eliminated many springs and seeps and increased the severity and frequency of stream scouring during storm events. However, the landscape at Acadia has not undergone such extensive land-use changes, and these mechanisms do not seem likely here.

3.2. *Potential causes of decline*

Effects of non-native fishes on lentic (Bradford, 1989; Bradford et al., 1993, 1994; Brönmark and Edenhamn, 1994; Braña et al., 1996; Hecnar and M'Closkey, 1997; Bradford et al., 1998; Goodsell and Kats, 1999; Knapp and Matthews, 2000; Pilliod and Peterson, 2001) and lotic (Gamradt and Kats, 1996; Resetarits, 1997) amphibian population dynamics are relatively well studied. Effects of freshwater invasive species on native biota may alter the behavior of native species primarily through competition or predation and negative effects can occur at multiple ecological scales including individual, population, community and ecosystem levels (Simon and Townsend, 2003). Salmonid introductions may cause trophic cascades resulting in increased algal biomass and production and may cause changes in nutrient flux in lotic environments (Simon and Townsend, 2003 and references therein). ANP native (i.e., stocked and natural populations) and non-native fish likely had the ability to reduce reproductive output of northern dusky salamander populations by direct pressure on eggs and larvae (i.e., life stage specific effects), although pressure on larvae was likely more important than pressure on eggs since adults may lay eggs in both in-stream habitats or in moist, riparian sites, often $\leqslant$50 cm from the active stream channel (Petranka, 1998; Hunter et al., 1999; Lowe et al., 2004). Predatory native and non-native fish may have altered the breeding behavior of adult northern dusky salamanders and prevented successful mating and oviposition, however further research is required to evaluate this hypothesis. Kats and Sih (1992) demonstrated that oviposition site selection of streamside salamanders (*Ambystoma barbouri*) was negatively influenced by native green sunfish (*Lepomis cyanellus*). However since little information is available on fish population dynamics in ANP stream ecosystems (although see Lê and Moring, 1999; Stone et al., 2001), it is difficult to understand the full extent to which fish potentially limited the occurrence, abundance and fitness of the northern dusky salamander at ANP.

Chytridiomycosis is an infectious disease that negatively affects amphibians and is caused by the chytrid fungus, *Batrachochytrium dendrobatidis* (Berger et al., 1998). The influence of chytrid fungus on the population dynamics and decline of the northern dusky salamander at ANP is unknown, although this fungus along with irridovirus, has been reported in tadpole populations from ANP (Bangor Daily News, 2002). Additionally, Cummer et al. (2005) reported the occurrence of the aquatic chytrid pathogen in a terrestrial plethodontid salamander. To our knowledge the occurrence of chytrid fungus has not yet been reported in the northern dusky salamander.

Streams in ANP have embedded substrates, episodic acidification events, and native (i.e., stocked and natural) and non-native fish populations can utilize these lotic habitats to travel between pond habitats within ANP watersheds. Additionally, atmospheric pollutants such as nitrate, sulfate, and Hg, commonly occur in these stream ecosystems and the cumulative negative effects from all of these stressors may be the best explanation for the observed decline of this population (Fig. 2). Gore (1983) reported that the combination of high acidity and high levels of conductivity (i.e., a pollution indicator metric) limited the occurrence of *Desmognathus* larval salamanders in streams affected by mines of the Cumberland Plateau in Tennessee.

ANP experiences scouring storm events and episodic pulses of acidity particularly in spring and fall. However, these weather patterns, storm events, and acidity pulses also occurred when northern dusky salamanders were historically observed. However, the magnitude of the acidity pulse from historic episodic events in comparison to current episodic events is unknown. There is no evidence that the 1947 fire at ANP negatively affected northern dusky salamander populations since there was no observed change in the minimum known percent of streams occupied before and after this historic disturbance event (Fig. 2).

Bank (2005) reported that *E. bislineata bislineata* larval occurrence and abundance was negatively affected by lower pH, increasing brook trout abundance, and by increases in percent of stream substrate embeddedness (quantified as the percent of individual cobbles buried in fine sediments such as silt or sand – Lowe and Bolger, 2002). Substrate embeddedness may have also negatively affected larvae of northern dusky salamanders through both top-down and bottom-up processes, however no data are available to evaluate temporal variability in substrate embeddedness at ANP. Fine sediment can reduce periphyton productivity and reduce invertebrate prey resources for salamanders (Ryan, 1991; Anderson, 1992; Rosenberg and Resh, 1993) and stage specific effects of sedimentation on stream salamanders have been previously reported (Lowe et al., 2004). Embedded substrates may also reduce the availability of interstitial areas, which often serve as a refuge from predators and may increase mortality of larvae and adults via direct or indirect effects of high flows (Hart and Finelli, 1999), fish predation (Sih et al., 1988; Sih and Kats, 1991) or the interaction between these stressors (Lowe and Bolger, 2002). Removal of streamside cover objects and the resulting siltation may degrade stream salamander habitat (Petranka et al., 1994). Morevoer, Wheeler et al. (2003) reported that stream dwelling hellbender (*Cryptobranchus alleganiensis alleganiensis* and *C. alleganiensis bishopi*) populations declined by ∼77%, over a 20-yr period, at five river sites in Missouri. Although the actual cause of the hellbender decline was determined to be unknown, the authors postulate that habitat degradation (i.e., siltation and eutrophication) and chemical pollution were likely, at least, partly responsible (Wheeler et al., 2003).

Fig. 1 illustrates the spatial juxtaposition of total Hg (ng/g wet wt.) in *E. bislineata bislineata* larvae across all sites where salamander collections were permitted by the NPS. Although there are only two stream sample sites on the west side of ANP that were analyzed for total Hg in *E. bislineata bislineata* larvae, both streams had high concentrations of total Hg. Moreover, Burgess (1997) similarly reported high levels of total Hg in fish from the Hodgdon and Seal Cove Ponds located directly to the west of these sampling sites (Fig. 1) from this investigation. These data combined suggest that the west side of ANP may have a higher Hg methylation potential and a greater Hg exposure to animal biota (especially animals with low vagility) inhabiting this sector of ANP.

Bank et al. (in press) reviewed Hg levels in biota from ANP and reported that organisms from all trophic positions inhabiting the different surface water types throughout ANP had elevated Hg levels. Bank et al. (2005) measured Hg in larval and adult *E. bislineata bislineata* salamanders in ANP headwater streams and reported that larvae and adults can often bioaccumulate total Hg, mostly (73–97%) in the highly toxic methylmercury form, at levels >100 ng/g (wet weight). Additionally, 22.7% of *E. bislineata bislineata* larvae, at ANP, had Hg levels that exceeded the methylmercury criterion $\geqslant$77 ng/g, for the protection of piscivorous wildlife (USEPA, 1997). Amphibians are likely at a higher risk of negative effects from Hg contamination than other species such as birds, mammals, and reptiles who can store Hg in body parts away from vital organs such as feathers, hair, or carapaces, respectively (Bank et al., in press). Moreover, Unrine et al. (2004) suggest that atmospheric deposition of Hg in aquatic ecosystems may have significant effects on the southern leopard frog (*Rana sphenocephala*) during the larval developmental stage and that Hg may increase incidence of death and malformations and affect growth regulation, overall fitness, and the timing of metamorphosis. Northern dusky salamanders foraging on larval and adult northern two-lined salamanders and other contaminated prey items may have experienced reduced fitness as a result of the chronic exposure to Hg and other contaminants in these food sources. Additionally, precipitated aluminum likely also negatively affected northern dusky salamanders since aluminum is highly toxic and can interfere with larval respiratory exchange (i.e., via accumulation on gills), cause reduced larval development, and disrupt the sodium balance in all life stages of aquatic biota (Haines, 1981; Brown and Sadler, 1989; Rosemond et al., 1992).

4. Conclusions

Our data indicate that the northern dusky salamander at ANP has experienced a significant population decline (Fig. 2). Although the actual causes of the population decline are unknown, this species likely declined as a response to the cumulative effects of multiple abiotic and biotic stressors. The potential stressors include fish stocking, substrate embeddedness, fungal pathogens, indirect and/or direct effects of atmospheric pollutants, or the interaction among identified stressors. Streams at ANP experience episodic acidification events that can cause enhanced leaching of toxic aluminum and Hg and, in general, at lower pH, heavy metals are more soluble which can create degraded habitat conditions for sensitive aquatic biota such as salamanders. Hg contamination is widespread in biota inhabiting the lakes, ponds, and streams at ANP, including species that served as prey for northern dusky salamander such as *E. bislineata bislineata* (Bank et al., 2005, Fig. 1). No direct information exists for toxicity of aluminum and Hg to northern dusky salamanders, however, Hg has no benefit to biota and is an extremely powerful neurotoxin and toxic effects can occur at extremely low levels (Eisler, 2000). Additionally, several authors have reported the negative effects of aluminum on amphibians (Clark and Hall, 1985; Clark and Lazerte, 1985; Freda and McDonald, 1990), fish (Haines, 1981; Brown and Sadler, 1989) and invertebrate species (Rosemond et al., 1992). Of the stressors we identified in this study, we believe that atmospheric pollutants, and the resulting processes related to the enhanced mobilization of toxic substances, from adjacent riparian, hydric soils, such as Hg and aluminum (Pellerin et al., 2002), are likely the best explanation for the decline of the northern dusky salamander at ANP. Therefore, Hg and other atmospherically based pollutants, and surface water contamination in general, should be considered as important, potential environmental stressors that may negatively affect amphibian populations. It is also important to mention that the influence of chytrid fungus on the population decline of northern dusky salamanders at ANP is currently unknown. We recommend the expansion of biomonitoring of amphibians along a gradient of human disturbance and land use to determine the relationships between amphibian distributions, population declines and habitat degradation (Bank, 2005). We also recommend that future studies evaluate the effects of Hg and heavy metal pollution on spatial and temporal patterns of biodiversity.

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