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Effects of extreme floods on trout populations and fish communities in a Catskill Mountain river

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SUMMARY

1. Extreme hydrologic events are becoming more common with changing climate. Although the impacts of winter and spring floods on lotic ecosystems have been well studied, the effects of summer floods are less well known.

2. The Upper Esopus Creek Basin in the Catskill Mountains, NY, experienced severe flooding from Tropical Storm Irene on 28 August 2011, and peak discharges exceeded the 0.01 annual exceedance probability (>100 year flood) in some reaches. Three years of fish community data from pre-flood surveys at nine sites were compared to data from 2 years of post-flood surveys to evaluate changes in fish communities and populations of brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*).

3. Basinwide, fish assemblages were not strongly impacted and appeared highly resilient to the effects of the flood. Total density and biomass of fish communities were greater at most sites 10–11 months after the flood than 1 month prior to the flood while richness and diversity were generally unchanged. Community composition did not differ significantly between years or between the pre- and post-flood periods.

4. Although the density of mature brown trout was low at most sites (mean density = 146 fish ha^{-1}), young-of-the-year brown trout reached their highest density (mean = 2312 fish ha^{-1}) during 2012. In contrast, rainbow trout densities declined substantially during the 5-year study and the 2012 year class was small (mean density = 222 fish ha^{-1}).

5. Late summer floods may be less damaging to stream fish communities than winter or spring floods as spawning activity is negligible and early life stages of many species are generally larger and less susceptible to displacement and mortality. Additionally, post-flood conditions may be advantageous for brown trout recruitment.

Keywords: brown trout, Esopus Creek, fish communities, flood, rainbow trout

Introduction

Catastrophic floods can seriously damage in-stream and riparian habitats and resident fish assemblages in mountain streams. Although lotic fish communities have evolved with dynamic geomorphological conditions and are relatively resilient to extreme hydrologic events (Nislow *et al.*, 2002; Lytle & Poff, 2004), severe floods may reduce fish density and biomass and influence community composition (Roghair, Dolloff & Underwood, 2002; Carline & McCullough, 2003; Warren, Ernst & Baldigo, 2009; Milner *et al.*, 2012). Direct effects involve displacement-related mortality and destruction of incubating eggs, while indirect effects to habitat can affect carrying capacity or favour one species or guild over others (Elwood & Waters, 1969). A better understanding of the short- and long-term impacts of floods on fish assemblages is necessary to identify factors that afford resistance or resilience. Since most climate change models predict an increased frequency of extreme hydrologic

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events (Fowler & Hennessy, 1995; Rosenzweig *et al.*, 2011), such information will become invaluable to manage fish communities and protect species of concern.

The factors that influence flood impacts on lotic fish communities include flood magnitude, the availability of suitable refuges (e.g. floodplains and backwaters) (Jowett & Richardson, 1989; Pearsons, Li & Lamberti, 1992; Lake, 2000) and flood timing relative to the life history of resident species (Harvey, 1987; Strange, Moyle & Foin, 1992; Fausch *et al.*, 2001). Bed-mobilising events that occur while eggs are in the gravel or elevated flows shortly after fry emerge are particularly detrimental to fish populations (Warren *et al.*, 2009). For salmonids, the risk of displacement during high flows is greatest when fry emerge from the gravel and enter the free-feeding stage where displacement downstream can occur by velocities as low as 0.1 m s^{-1} (Heggenes & Traaen, 1988). This is known as the critical period because the size of the 0 + year class [herein termed young-of-the-year (YOY)] determines the initial cohort size and limits the future strength of this year class (Nislow, Einum & Folt, 2004). The strength of salmonid year classes has often been correlated with hydrologic conditions that occur at the time of emergence (Jensen & Johnsen, 1999; Spina, 2001; Cattanéo *et al.*, 2002; Lobon-Cervia, 2004).

Winter floods can adversely affect the eggs of autumn-spawning salmonids, such as brown trout (*Salmo trutta*) and brook trout (*Salvelinus fontinalis*), thereby providing a potential competitive advantage to spring-spawning species, such as rainbow trout (*Oncorhynchus mykiss*) (Strange *et al.*, 1992; Warren *et al.*, 2009). However, late spring floods that occur well after the fry of autumn spawners have emerged and advanced beyond their critical period can scour the eggs or kill newly emerged fry of spring spawners (Seegrist & Gard, 1972). Few studies, however, have documented the effects of summer floods on fish assemblages (Bischoff & Wolter, 2001; Jurajda, Reichard & Smith, 2006), in particular on sympatric trout species (Pearsons *et al.*, 1992; Nislow *et al.*, 2002). In the north-eastern United States, large summer floods have occurred less frequently than spring snowmelt-driven floods, so resident species may be poorly adapted for and more sensitive to summer floods (Giller, 1996; Jones & Petreman, 2013).

In this paper, we quantify the impact and recovery of stream fish communities to catastrophic summer floods in mountain streams in New York State (Fig. 1). Fish communities were previously surveyed annually at 18 mainstem and tributary sites within the Upper Esopus Creek (herein termed Esopus) Basin in the Catskill Mountains in south-eastern New York from 2009 to 2011, as part of a study to evaluate the effects of supplemental flows from an inter-basin aqueduct (Shandaken Tunnel). Approximately 1 week after the conclusion of the 2011 surveys, the region experienced severe flooding from Tropical Storm Irene (Fig. 2). Rainfall at Slide Mountain in the headwaters of the Esopus totalled 29.3 cm between 28 and 29 August 2011 during this event (Lumia, Firda & Smith, 2014). The annual exceedance probabilities (AEP) for peak flows at five

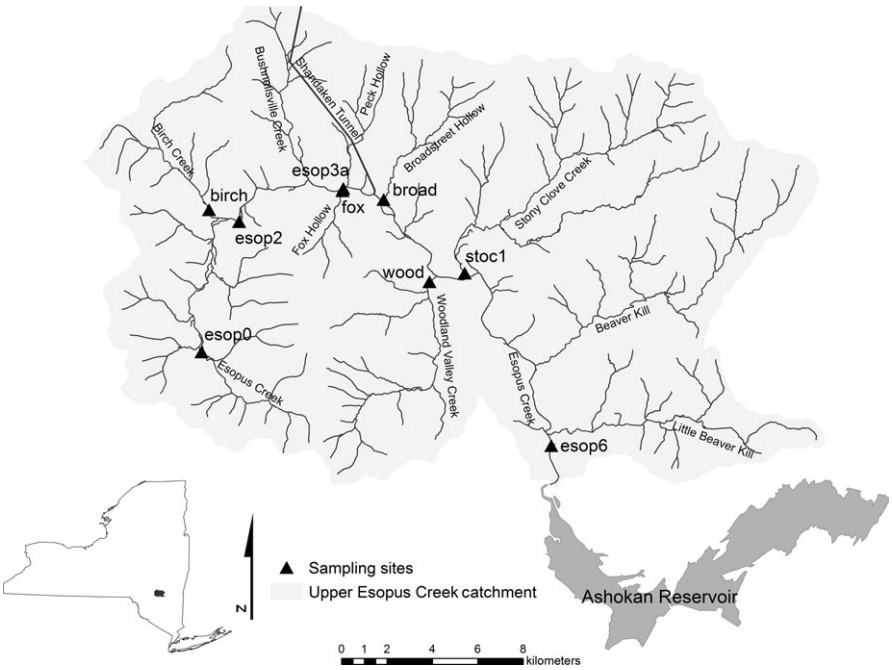

Fig. 1 Map showing nine sites on the Upper Esopus Creek and tributaries where fish communities were sampled from 2009 to 2013.

Table 1 Stream and study site name, site ID, drainage area (DA, km^2), elevation (m), peak discharge (m^3 s^{-1}), annual exceedance probability and recurrence interval (years) for the flood that occurred on 28 August 2011 in the Upper Esopus Creek Basin (Lumia *et al.*, 2014, Lumia, Firda & Smith, 2014)

Stream and site name	Site ID	DA (km^2)	Elevation (m)	Peak discharge (m^3 s^{-1})	Annual exceedance probability	Recurrence interval (years)
Fox Hollow	fox	10.3	309.4	–	0.067*	15*
Broadstreet Hollow	broad	23.7	295.8	–	0.015†	65†
Birch Creek	birch	32.4	377.4	41.3	0.143	7
Woodland Valley Creek	wood	53.4	267.6	189.4	0.067	15
Stony Clove Creek	stoc1	83.9	245.2	404.9‡	0.012‡	80‡
Esopus Creek at Oliverea	esop0	30.3	454.5	–	0.008§	>100§
Esopus Creek at Big Indian	esop2	111.9	354.9	–	0.008§	>100§
Esopus Creek at Allaben	esop3a	165.0	304.6	829.7	0.008	>100
Esopus Creek at Boiceville	esop6	497.3	188.8	2146.4	0.014	70

*Ungaged site; AEP and recurrence interval taken from USGS stream gage at wood.
†Ungaged site; AEP and recurrence interval taken as averages from stream gages on adjacent streams (Stony Clove Creek and Bushnellsville Creek).
‡Data from a USGS stream gage located on Stony Clove Creek 3 km upstream of stoc1.
§Ungaged site; AEP and recurrence interval taken from USGS stream gage at esop3a.

permanent U.S. Geological Survey (USGS) stream gages in the basin ranged from 0.143 to 0.008 (Lumia *et al.*, 2014), which corresponded to flood recurrence intervals of 7 to >100 years (Table 1). Discharge had nearly receded to pre-flood levels by 7 September 2011, at which point a moderate flood from the remnants of Tropical Storm Lee affected the basin again. The AEP for peak flows during this flood varied between >0.500 and 0.333 (recurrence intervals of <2 to 3 years) at the five stream gages. In-stream and riparian habitats were severely altered at most study sites, and emergency repairs by local municipalities and landowners further affected several study reaches. Post-flood surveys of fish assemblages were conducted during 2012 and 2013 to increase our understanding of the response and recovery of fish assemblages to floods. Three hypotheses were tested: (i) there would be a basinwide decline in fish community metrics; (ii) fish community composition would be altered; and (iii) YOY trout would be more affected than older life stages.

Methods

Study area

The Upper Esopus Creek is in the south central Catskill Mountains of south-eastern New York, USA (Fig. 1). The Esopus follows a 41.8 km course from its headwaters to the Ashokan Reservoir. The 497.3 km^2 catchment is over 95% forested and includes lacustrine–clay deposits that contribute suspended sediments to the system (CCE, 2007). Flows in the Esopus are supplemented by inputs from the Schoharie Reservoir (in the Mohawk River drainage) via the Shandaken Tunnel, a 29-km aqueduct joining the Esopus near Shandaken, NY, *c.* 23 km downstream from its headwaters. Fish assemblages were surveyed at nine of the original 18 study sites during the summers of 2012 and 2013. Four of these sites were located on the mainstem of the Esopus, and five were located on major tributaries near their confluence with the Esopus (Table 1).

Fish surveys

Two similar fish survey techniques were used to quantify fish assemblages at small- and large-channel study sites (reaches). Reach lengths were 10–20 times mean channel widths and typically encompassed one or two complete geomorphic channel-unit sequences (Simonson, Lyons & Kanehl, 1994; Fitzpatrick *et al.*, 1998; Meador, McIntyre & Pollack, 2003). Fish were collected with a backpack electrofisher using a three-pass depletion method, and three to five people netted fish. In narrow channels (<15 m), blocking seine nets were placed completely across the channel at the upstream and downstream end of study reaches. At sites with wide channels (>15 m), three (replicate) surveys were conducted in relatively small near-shore subreaches. At each subreach, one blocking seine was affixed to the bank and then stretched perpendicular to the bank and attached to a rock or rebar (6–8 m from shore); a second 25-m seine was oriented parallel to shore and then attached to a second rock or rebar (also 6–8 m from shore); and a third seine was placed between the second rebar and shore. All fish were identified to species and the length and weight of fish >150 mm long were

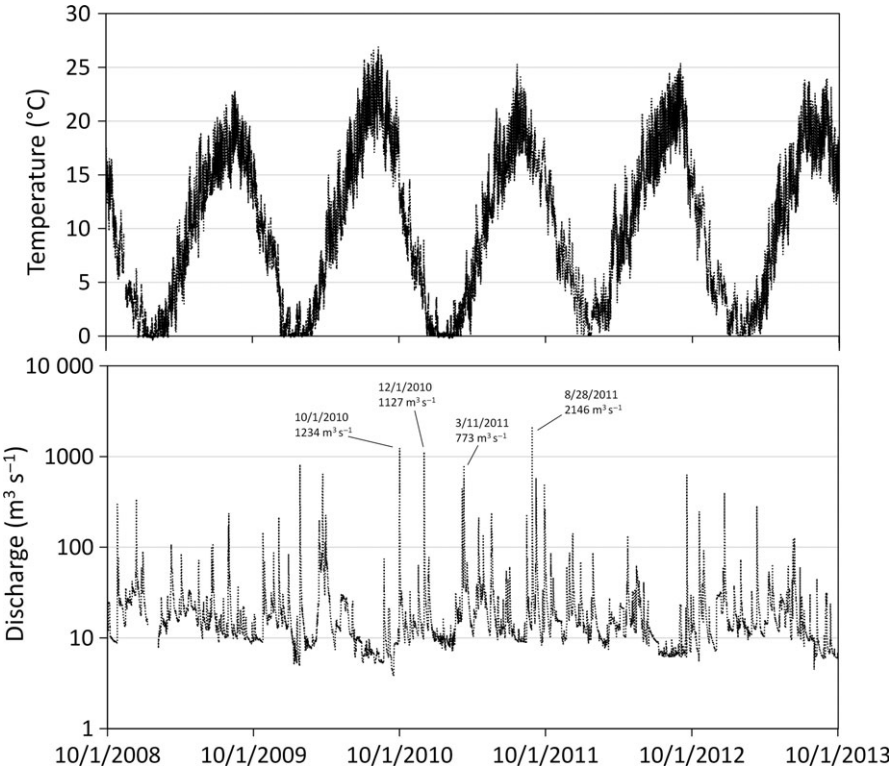

Fig. 2 Stream temperature and discharge at esop6 every 15 min from 1 October 2008 to 1 October 2013.

recorded. The length and weight of some small abundant species were obtained from 30 individuals, after which pooled weights were recorded by species in batches of up to 25 fish. All fish were subsequently returned to the stream. Surveys were conducted during summer base flow periods.

Within each reach and subreach, the total reach length and widths of 10 evenly spaced transects were recorded. A modified point-and-transect method (Fitzpatrick *et al.*, 1998) was used to measure depth and velocity and to estimate dominant-substrate size categories at three points (at centre, 25 and 75% of each cross-section) along each of the 10 transects. The total length and mean width were used to calculate reach area, and mean reach depth and velocity were determined using all transect values.

Data analysis

The number of fish captured during each pass was used to estimate population size and biomass (and 95% confidence intervals for each) for the community and for each species' population at each site using a maximum-likelihood population estimator built on inherent assumptions in the Moran-Zippin method of proportional reduction (Zippin, 1958; Van Deventer & Platts, 1983, 1985). These values were divided by the total area sampled at each study site to estimate density and biomass of fish per hectare. Two components of diversity, total species richness and Simpson's diversity index (D, reinterpreted as $1–D$) were also estimated for each site (Simpson, 1949; Whittaker, 1975). Mean metric scores were compared between years using one-way repeated-measures analysis of variance (rANOVA) with Fisher's LSD *post hoc* test and between pre- and post-flood periods using paired *t*-tests.

The composition of fish assemblages was assessed at the community and taxonomic group levels. The most abundant fish were grouped by family as Salmonidae (*S. trutta*, *O. mykiss*, *S. fontinalis*), Cottidae (*Cottus cognatus*), Cyprinidae (*Rhinichthys atratulus*, *R. cataractae*, *Pimephales promelas*, *P. notatus*, *Exoglossum maxillingua*, *Semotilus corporalis*, *S. atromaculatus*, *Notemigonus crysoleucas* and *Luxilus cornutus*) or Catostomidae (*Catostomus commersonii* and *C. catostomus*) (herein referred to by the common names trout, sculpin, minnow and sucker), and the remaining 11 uncommon species were grouped as 'other'. Additional analysis of spatial patterns in fish community composition was conducted using multivariate techniques with PRIMER-E V6 software. Square-root-transformed species densities were used to form a resemblance matrix of Bray–Curtis similarities comparing all samples, and analysis of similarities (ANOSIM) and a non-metric multidimensional scaling (MDS)

ordination were used to test the hypothesis that species assemblages differed between pre-flood and post-flood surveys (Clarke & Warwick, 2001; Clarke & Gorley, 2006).

Changes in brown trout and rainbow trout populations were assessed using both raw counts and the density and biomass of each population. Brook trout were not included in this analysis due to their low abundance. Length–frequency distributions for brown and rainbow trout were created for each year by pooling all individuals across sites and were used to estimate the percentage of individuals that were YOY based on length (<101 mm for brown trout and <91 mm for rainbow trout). Resultant percentages and the estimated total density for each species were used to determine the estimated densities of YOY and older fish for each species and year.

Results

Fish community metrics

Each community metric increased between the pre-flood and post-flood year (Fig. 3), although mean species richness ($P = 0.422$, rANOVA) and species diversity ($P = 0.299$) did not differ significantly between years. Mean community density differed significantly between years ($P = 0.006$), and pairwise comparisons showed that 2011 was significantly lower than 2009 ($P < 0.001$, Fisher's LSD test), 2012 ($P = 0.012$) and 2013 ($P = 0.024$) and that 2009 differed from 2010 ($P = 0.044$). Mean community biomass also differed significantly between years ($P = 0.042$), and pairwise comparisons showed that 2011 differed significantly from 2009 ($P = 0.003$) and 2012 ($P = 0.035$). None of the four metrics differed significantly between the pre- and post-flood periods ($P > 0.05$, paired *t*-test). At individual sites, annual differences in community density and biomass (see Fig. S1) and richness and diversity (not shown) generally mirrored the mean data (Fig. 3) and are not explored further herein.

Fish community structure

The three taxonomic groups with the greatest mean density were minnow, sculpin and trout, respectively, and this order was maintained throughout the study (Fig. 4a). Mean density of these three dominant groups declined together from 2009 to 2011 but then increased in 2012 and diverged in 2013. The mean biomass of trout was greatest each year, minnow and sculpin were typically 50–75% of trout biomass, and sucker was highly variable (Fig. 4b). Mean biomass of the four dominant groups declined from 2009 to 2011, increased in 2012 and either changed negligibly (minnow and sculpin), increased (trout) or decreased (sucker) in 2013.

The wide dispersion of sites across both axes of the MDS ordination (Fig. 5) indicates fish assemblages differed considerably between sites, but not between pre-flood (2009–2011) and post-flood (2012–2013) surveys. ANOSIM confirmed no significant differences between fish assemblages sampled in different years (global $R = 0.018$, $P = 0.273$) nor were pre-flood (2009–2011) communities significantly different from post-flood (2012–2013) communities (global $R = -0.004$, $P = 0.456$). Pairwise comparisons showed the assemblages from 2009 and 2011 were least similar, while assemblages sampled during 2010 and 2012 were the most similar. Sites with small drainage areas and low species richness, such as fox and esop0, clustered to the left of the ordination, whereas mainstem sites with the largest drainage areas, such as esop3a and esop6, generally grouped to the right.

Trout populations

Density and biomass of brown trout generally followed the pattern exhibited by the overall fish community. Mean brown trout density decreased from 2009 to 2011, increased sharply in 2012 and decreased in 2013 (Table 2). Mean biomass also decreased between 2009 and 2011 but then increased in both 2012 and 2013. Mean weight of individual brown trout was highest in 2011, lowest in 2012 and intermediate in other years.

Temporal variation in density and biomass of rainbow trout followed a different pattern. Mean rainbow trout density was similar in 2009 and 2010, but then declined to its lowest level by 2013 (Table 2). Biomass of rainbow trout increased between 2009 and 2012 but then also declined to its lowest level in 2013. Unlike brown trout, mean body weight of rainbow trout increased each year of the study.

The strength of various size classes for brown trout and rainbow trout populations varied considerably between years and differed from each other within the same years (Fig. 6). Small (YOY and yearling) brown trout were abundant during 2009, decreased in 2010 and declined to their lowest abundance during 2011, immediately before the flood, and then rose sharply during 2012, the first year after the flood. Substantial increases in YOY brown trout were observed between 2011 and 2012 at some sites. Most notably, no YOY brown trout were sampled from the fox study reach during 2011, yet

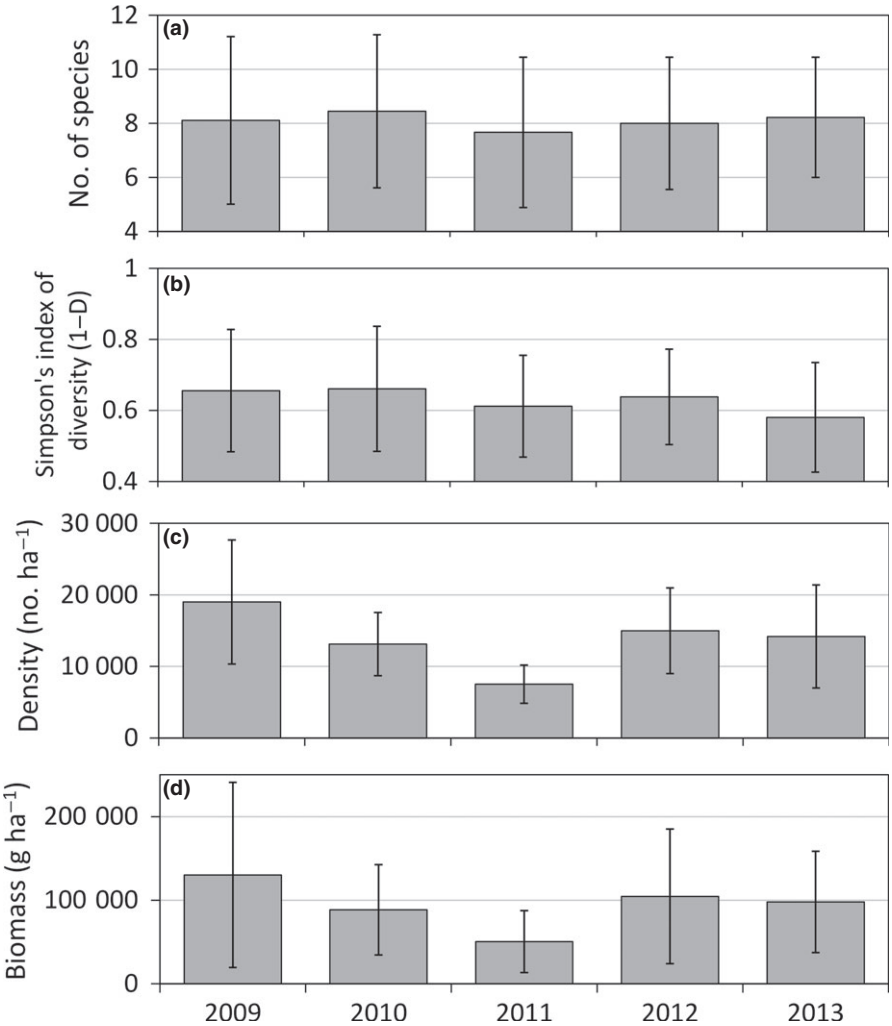

Fig. 3 Mean (n = 9) richness (a), diversity (b), density (c) and biomass (d) for 2009–2013. Whiskers show 1 SD about the mean.

they existed at a density of 12 860 fish ha^{-1} during the 2012 survey (see Appendix S1). During the following year (2013), the numbers of brown trout YOY decreased and the numbers of yearlings increased, reflecting a relatively well-balanced age structure not seen since 2009 (Fig. 6). The decrease in density and change in age structure of rainbow trout were similar to that observed for brown trout from 2009 to 2011, but the population did not recover in 2012 and 2013 (Fig. 6, Table 2). Only fox and esop3a displayed large increases in YOY rainbow trout between 2011 and 2012 as observed with brown trout populations at many sites.

Discussion

The variables determining the short-term response of fish assemblages to extreme floods in the Esopus appear complex. The magnitude of the flood (Resh *et al.*, 1988) and the availability of refuges (e.g. floodplains and backwaters) (Jowett & Richardson, 1989; Pearsons *et al.*, 1992; Lake, 2000) are two key variables that should control the severity of flood impacts. Although flood magnitude varied widely across sites in the Esopus Basin, peak discharges at many sites exceeded the 0.0167 AEP discharge (Table 1), comparable or greater to that observed in other studies that detected flood-related fishery impacts (Jowett & Richardson, 1989; Carline & McCullough, 2003). In addition, channels at most study sites were highly incised or adjacent to roads and essentially disconnected from their floodplain, the primary refuge area for fish during major flooding (Ross & Baker, 1983; Schwartz & Herricks, 2005). Despite this limitation, fish communities in the Esopus were relatively unaffected. Indeed, the density and biomass of fish communities at most sites were significantly higher 10–11 months after the flood than immediately prior, enabling us to reject our first hypothesis. The lack of severe community impacts was surprising given the flood magnitude and poor floodplain connectivity and suggests others factors increased the resistance or resilience of fish assemblages.

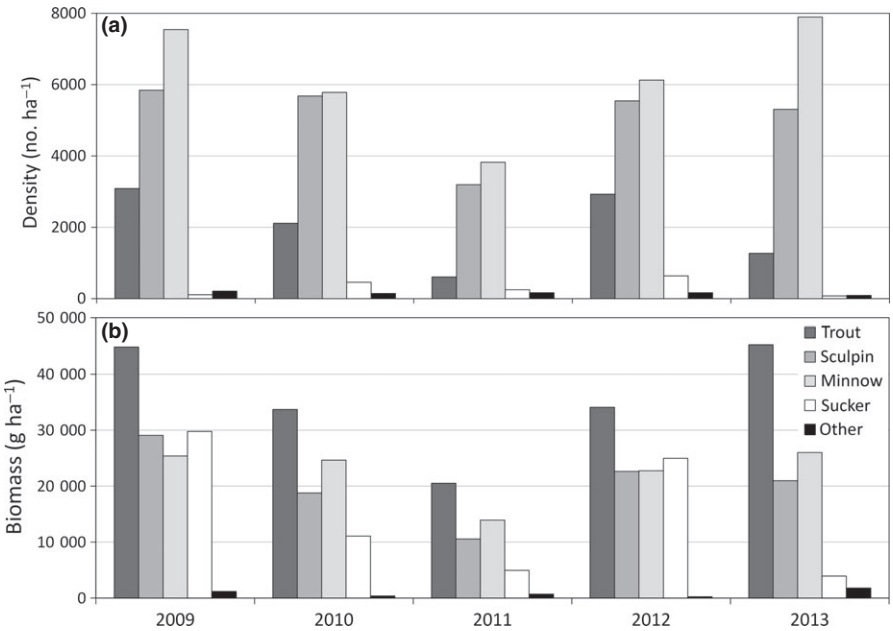

Fig. 4 Mean density (a) and biomass (b) for each taxonomic group during each year.

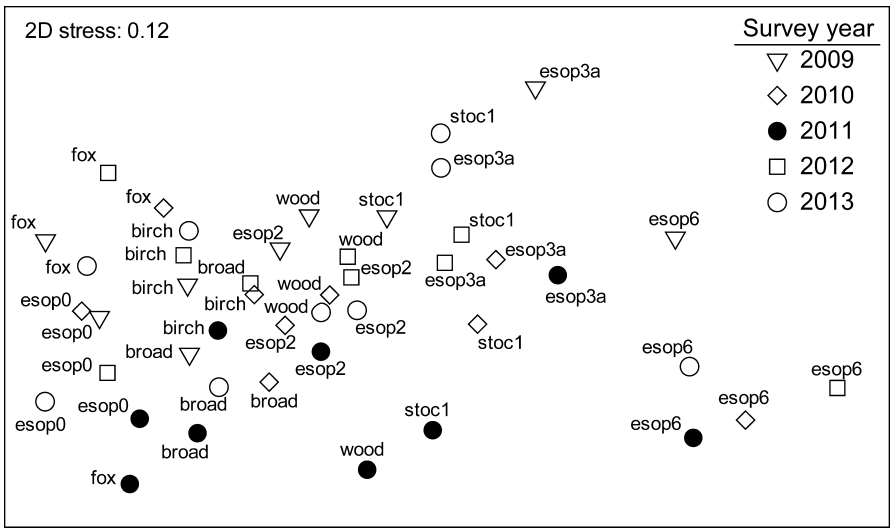

Fig. 5 Non-metric multidimensional scaling ordination showing fish assemblages at each site during each year.

There is increasing evidence that the timing of hydrologic events can affect the severity and nature of their ecological impacts (Harvey, 1987; Pearsons *et al.*, 1992; Giller, 1996; Fausch *et al.*, 2001) and the timing of the late summer flood triggered by Irene may partly explain why larger impacts to the fish community were not observed. First, flooding from Irene did not coincide with spawning or fry emergence of most resident species. For example, YOY brown trout and rainbow trout in the Esopus emerge as swim-up fry in the spring at which time they are *c.* 20 mm long (Carlander, 1969) and highly susceptible to displacement from elevated flows. By late August, they reach lengths of 60–90 mm and 40–70 mm, respectively, and can hold their positions against elevated velocities (Heggenes & Traaen, 1988). Although much smaller than age 0 + trout, the YOY of other abundant species such as slimy sculpin (*C. cognatus*) and most cyprinids would have also advanced beyond the critical period by late August (Harvey, 1987). Additionally, as poikilotherms, fish metabolism is higher during warm periods, suggesting the ability to withstand greater velocities and thus resist displacement (caused by floods) more effectively than during cold periods (Glova & McInerney, 1977; Heggenes & Traaen, 1988). Accordingly, more severe impacts would be expected from a flood of similar magnitude occurring during winter or spring.

Table 2 Characteristics of brown trout and rainbow trout populations at all nine study sites by year

Year	Mean density (no. ha^{-1})	Percentage of YOY (%)	Density of YOY (no. ha^{-1})	Density of non-YOY (no. ha^{-1})	Mean biomass (g ha^{-1})	Mean individual weight (g)
Brown trout						
2009	1903	73	1382	521	39 498	21
2010	896	39	352	544	27 518	37
2011	225	49	111	115	11 612	48
2012	2457	94	2312	146	22 545	9
2013	1049	51	530	519	39 854	37
Rainbow trout						
2009	1137	90	1021	117	4909	5
2010	1178	80	946	231	5899	6
2011	378	57	214	164	8756	23
2012	429	52	222	207	11 385	26
2013	165	53	87	78	4813	30

YOY, young-of-the-year.

The unusually poor status of the Esopus fish communities during 2011 (before the late summer flood) complicated the analysis of flood impacts and recovery. Most analyses indicated that density, biomass and composition of fish communities differed most or were at their lowest levels during 2011. A combination of unusual hydrologic and thermal conditions during 2010 and 2011 may have been responsible. A prolonged period of low flows occurred during the summer of 2010. Mean daily discharge at esop6 from June to September 2010 was 8.8 m^3 s^{-1}, or about 65% of the historical average of 13.6 m^3 s^{-1} for the same period (USGS, 2011a). The discharge at all other study sites that were not supplemented by waters from the Shandaken Tunnel departed even further from historical averages; for example, mean daily discharge at wood (0.4 m^3 s^{-1}) during this period was 29% of the long-term average (1.4 m^3 s^{-1}; USGS, 2011b). These extremely low flows coincided with an unusually long period (16 July to 20 August 2010) of warm stream temperature (e.g. mean 22 °C, maximum 26.9 °C at esop6) (Fig. 2) (USGS, 2011a), which approached or exceeded mean and maximum thermal tolerance limits for brown trout (Wehrly, Wang & Mitro, 2007). However, the 2010 fish surveys in the Esopus were conducted between late June and early August and may have been too early to detect the full effects of the drought. Additionally, several large floods affected the basin between the 2010 and 2011 surveys. The second, third and fifth largest floods observed at esop6 during the 5-year study took place on 1 October 2010, 1 December 2010 and 11 March 2011, respectively (Fig. 2). Therefore, any effects of the drought or cold-water floods on fish assemblages would have been most evident during the 2011 surveys and the changes in most metrics between 2011 and 2012, which might have been attributed solely to the effects from Irene, should more accurately be described as a response to the August 2011 floods and recovery from the 2010 drought and winter 2011 floods.

The large increase in the number of YOY brown trout during 2012 suggests that the species was resilient to the extreme summer flood. We did not investigate the cause for this increase, but suggest that the increase in YOY density could be a result of favourable spawning conditions during autumn 2011 and/or increased survival of eggs and fry during incubation and emergence. First, post-flood changes within stream channels may have exposed more trout spawning habitat or increased the quality of existing spawning habitat. Major floods can flush detrimental fine sediment from streams and generally coarsen the substrate (Jowett & Richardson, 1989; Carline & McCullough, 2003; Ortlepp & Mürle, 2003; Herbst & Cooper, 2010). This flushing is pertinent to the Esopus because many reaches bisect lacustrine–clay deposits and carry high loads of suspended sediment (CCE, 2007). Thus, the 2011 floods may have excavated fines from depositional areas and created larger- and higher-quality gravel beds, improving the survival of overwintering trout eggs during winter 2011–2012. Second, stream flows during winter–spring 2011–2012 were moderate and relatively stable, ideal for incubating embryos and emerging fry (Lobon-Cervia, 2004). Third, the low density of most species in summer 2011 suggests that the 2012 year class may have experienced less predatory and competitive pressure. Fourth, all study sites are located within 2.6–35.4 km of the 34 km^2 Ashokan Reservoir (Fig. 1). Brown trout from the Ashokan Reservoir spawn in the Esopus and

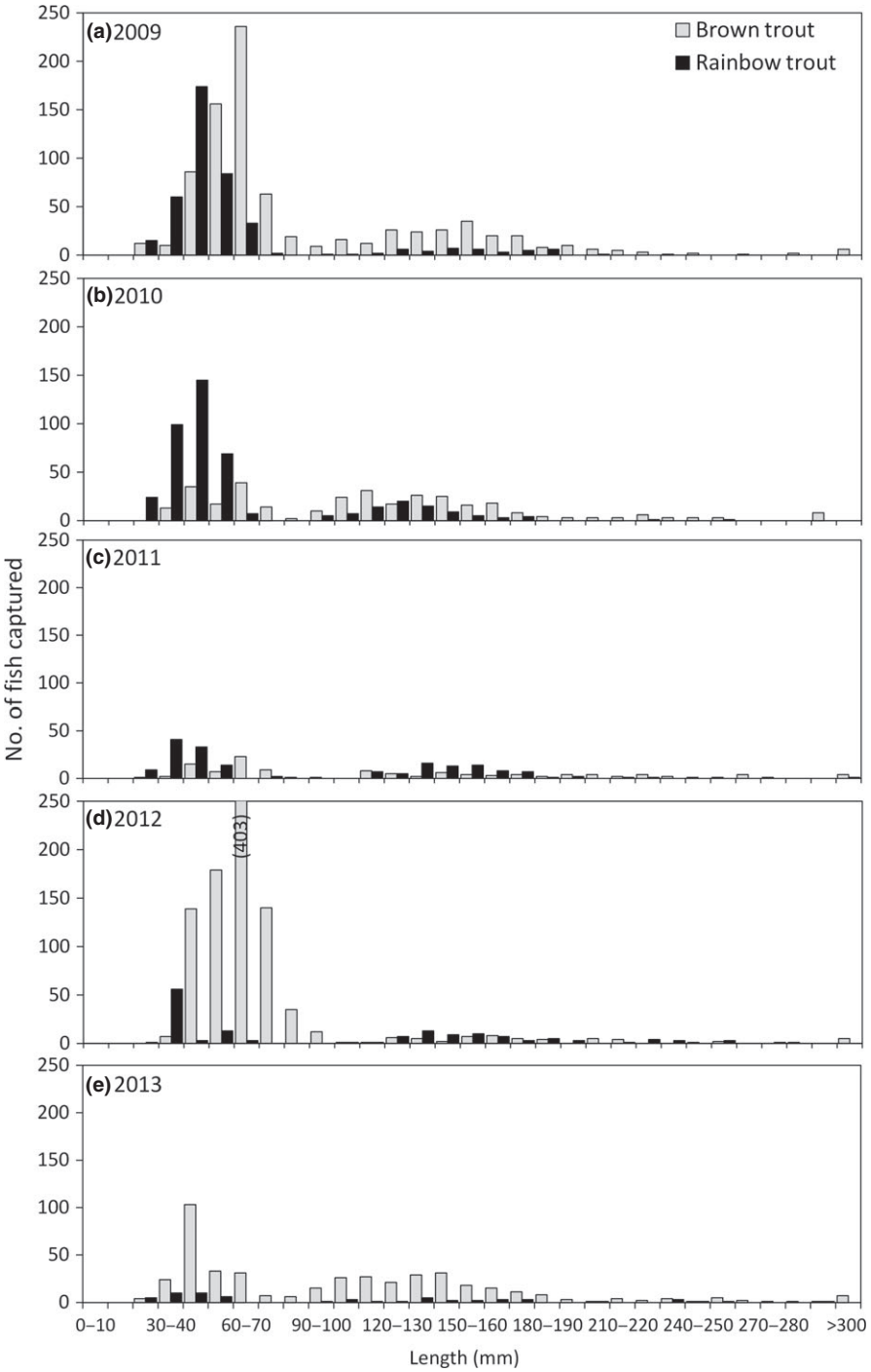

Fig. 6 Length–frequency distributions for brown trout and rainbow trout captured from all nine study sites during 2009–2013.

its tributaries every autumn. Because it is unlikely that the reservoir population of brown trout was severely affected by Irene, it would have provided a close source of spawners even if resident stream spawners had been eliminated from some reaches by autumn 2011. Together, these factors could have increased brown trout recruitment to the 2012 year class. Although brown trout spawning strategies appeared to promote resilience to the summer 2011 flood, the opposite would be expected if comparable floods occurred between November and April.

Unlike brown trout that exhibited a marked increase in YOY following the flood, the density of YOY rainbow trout was essentially unchanged between 2011 and 2012 (Table 2). The divergent responses of brown trout and rainbow trout populations to the summer flood could be explained by competitive interactions. Brown trout are autumn spawners, whereas rainbow trout are spring

spawners, and thus, YOY brown trout emerge earlier, are larger and may outcompete YOY rainbow trout (Gatz, Sale & Loar, 1987; Strange *et al.*, 1992). Conversely, the autumn spawning of brown trout increases the risk of egg loss if autumn or winter floods occur. Several investigations showed that YOY brown trout or brook trout were generally more numerous than YOY rainbow trout except during years when floods scoured the eggs of the autumn spawners (Seegrist & Gard, 1972; Strange *et al.*, 1992; Warren *et al.*, 2009). Thus, the summer 2011 flood and other factors may have increased the size of the brown trout year class during 2012, which may be partially responsible for poor rainbow trout recruitment during 2012 and 2013. Additional monitoring of rainbow trout populations in the Esopus, however, will be needed to further assess this relationship and the sustainability of these populations.

Our study has implications for assessing and managing natural resources in the Esopus and other streams of the region. First and most notably, it underscores the value of long-term monitoring. Our analyses were possible only because three consecutive years of baseline fish community data were available prior to the flood. The pre-existing fish community metrics placed the post-flood results into perspective and showed that the effects of the summer 2011 flood were relatively minor at the basin scale. Second, the fishery data provide information needed to gauge the ecological impacts of, and recovery from, both natural and anthropogenic disturbances in the future. These are valuable reference data sets, not only locally, but also regionally, because precipitation, temperature and stream discharge are expected to increase throughout the north-eastern USA (Fowler & Hennessy, 1995; Rosenzweig *et al.*, 2011).

Our 3 years of pre-flood data enabled us to test and reject two of the three proposed hypotheses. First, our results show that 10–11 months after the flood, fish community metrics were not adversely affected basinwide. Second, the post-flood composition of fish communities was not significantly different from the pre-flood condition. The low density of YOY brown trout and rainbow trout in 2011 prior to the flood made it difficult to test the third hypothesis of a disproportionate impact on this year class. Despite this, the persistence of yearlings of both species in 2012 suggests that this year class was not entirely eliminated and may not have even experienced unusually high mortality. Additionally, the high density of YOY brown trout in 2012 is one of our most important results and suggests excellent recruitment immediately following the flood. Together, these findings demonstrate that resident fish species in the Upper Esopus Creek were relatively resilient to effects of the extreme 2011 floods caused by Irene. Although additional research will be needed to validate these conclusions, our study indicates that resident fish species and their communities may be able to resist or recover rapidly from extreme floods.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Metrics for fish communities and populations of brown trout and rainbow trout for each study site surveyed, 2009–2013.

Figure S1. Density (a) and biomass (b) of fish communities at each site in each year.

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