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

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Effects of Extreme Hydrologic Events on Trout Populations
and Fish Communities in a Catskill Mountain River

by

Scott D. George

A Thesis
Submitted to the University at Albany, State University of New York
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Abstract

Extreme hydrologic events are becoming more common with changing climatic conditions. Although the impacts of winter and spring floods on lotic ecosystems have been well studied, the effects of summer floods on fish assemblages in mountain streams are less well known. This study took advantage of an extreme precipitation event during the course of an ongoing monitoring program, which allowed the opportunity to quantify the effects of a severe late summer flood on fish communities in a montane watershed. We also evaluated the effect of the flood on the density and age structure of two ecologically important populations of sport fish (brown trout *Salmo trutta* and rainbow trout *Oncorhynchus mykiss)*.

The Upper Esopus Creek Basin in the Catskill Mountains of New York underwent severe flooding from Tropical Storm Irene on August 28, 2011, with peak discharges that exceeded the 0.01 annual exceedance probability (>100 year recurrence interval flood) in some reaches. Three years of pre-flood fish community surveys, taken at nine sites, were compared with two years of post-flood surveys to evaluate changes in fish communities and populations of brown trout and rainbow trout.

Basin-wide, fish assemblages were not adversely affected and appeared to be highly resistant or resilient to any detrimental effects of the flood. Total density and biomass of fish communities were greater at most sites 10-11 months after the flood than one month prior to the flood while richness and diversity were generally unchanged. Community composition also did not differ significantly between years or between the pre- and post-flood periods. The density of mature brown trout was low at most sites
following the flood (mean = 15 fish 0.1ha\(^{-1}\)), while young-of-the-year (YOY) brown trout reached their highest density (mean = 231 fish 0.1ha\(^{-1}\)). In contrast, rainbow trout had a poor post-flood year class (mean YOY density = 22 fish 0.1ha\(^{-1}\)) and the overall population density of this species declined substantially during the five year study.

These findings suggest that late summer floods may be less damaging to stream fish communities than winter or spring floods. An important difference is that spawning activity occurs outside the flood period, so the early life stages of many species are generally larger and therefore less susceptible to displacement and mortality. The high density of YOY brown trout in 2012 following the flood suggests that post-flood conditions may have been advantageous for brown trout recruitment. Differences in life history between brown trout and rainbow trout and interspecific competition may be responsible for the divergent responses of these species.

The unusually poor condition of the fish community in 2011 immediately prior to the flood was unexpected and complicated the analysis of flood impacts and recovery. However, this finding suggests that other factors, such as drought and warm water temperatures during late 2010 and cold-water floods during winter 2011-2012, disturbed the fish community prior to the flood. Consequently, the increase in some metrics following the flood may be interpreted more as a recovery from this abnormal state than as a positive response to the flood. This study stands out as one of the few to examine effects of summer flooding on fish communities in montane streams and indicates that even extreme flood events during this period may not constitute a major disturbance to fish communities.
Acknowledgements

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# Table of Contents

Abstract.................................................................................................................................................. ii
Acknowledgements................................................................................................................................. iv
Figures...................................................................................................................................................... vi
Tables and Appendices ............................................................................................................................ viii
Introduction............................................................................................................................................... 1
    Study Area ........................................................................................................................................... 6
Methods.................................................................................................................................................... 6
    Hydrology ........................................................................................................................................... 6
    Fish Surveys ....................................................................................................................................... 7
    Data Analysis ...................................................................................................................................... 8
Results..................................................................................................................................................... 10
    Hydrology .......................................................................................................................................... 10
    Fish Community Metrics .................................................................................................................... 11
    Fish Community Structure .................................................................................................................. 14
    Trout Populations ................................................................................................................................. 16
    Relationships between Landscape/Hydrologic Variables and Fish Response.................. 20
Discussion............................................................................................................................................... 21
Conclusions............................................................................................................................................. 30
References............................................................................................................................................... 33
Figures

Figure 1 - Map showing the locations of study sites on the Upper Esopus Creek and tributaries and the years in which they were surveyed, 2009-2013.................................3

Figure 2 - The confluence of the Upper Esopus Creek and the Shandaken Tunnel in August, 2009 prior to Irene (left) and in September, 2011 less than one month after Irene (right)..................................................................................................................5

Figure 3 - Stacked bar graphs showing mean (n = 9) (a) richness, (b) diversity (1-D), (c) density (fish 0.1ha⁻¹), and (d) biomass (g 0.1ha⁻¹) for 2009-2013. Whiskers show one standard deviation about the mean........................................................................................................12

Figure 4 - Stacked bar graphs showing (a) density and (b) biomass of fish communities at each site for each year. Sites are arranged by drainage area and whiskers show the 95% confidence intervals based on depletion population estimates for each year.........13

Figure 5 - Stacked bar graphs showing mean (n = 9) (a) density and (b) biomass for each species’ group during each year ........................................................................................................15

Figure 6 - Non-metric multidimensional scaling ordination showing fish assemblages at each site during each year ........................................................................................................16

Figure 7 - Length frequency distributions showing the lengths of all brown trout and rainbow trout captured from nine study sites during 2009-2013.................................19
Figure 8 - Line graph showing unit (15-minute) values for stream temperature (°C) at esop6 from 10/1/2008-10/1/2013..........................25

Figure 9 - Line graph showing unit (15-minute) values for discharge (m³ s⁻¹) at esop6 from 10/1/2008-10/1/2013.................................................................28
Tables and Appendices

Table I - Stream and study site name, site ID, drainage area (km²), elevation (m), peak discharge (m³ s⁻¹), annual exceedance probability, and recurrence interval (years) for the flood that occurred on August 28, 2011 in the Upper Esopus Creek Basin............4

Table II - Mean (n = 9) density and biomass estimates from 3-pass depletion methods, percentage young-of-the-year (YOY) and mean weight from raw data of all sampled trout in each year irrespective of site, and YOY density and non-YOY density derived from estimated populations densities and percentage YOY of brown trout and rainbow trout..........................................................17

Table III - Spearman correlations (r) and P-values between change in fish metrics (2012-2011) with elevation, drainage area, and annual exceedance probability (AEP) for all nine sites (n = 9). Asterisk denotes a significant (P < 0.05) correlation .............21

Appendix I - Metrics for fish communities and populations of brown trout and rainbow trout for each study site surveyed, 2009-2013..........................................................39
Introduction

Catastrophic flooding events can seriously degrade in-channel and riparian habitats and associated resident fish assemblages in mountain streams and rivers. Although lotic fish communities have co-evolved with dynamic geomorphological conditions and are relatively resilient to extreme hydrologic events (Lytle & Poff, 2004; Nislow et al., 2002), significant reductions in the density and biomass of populations and shifts in fish community composition have been documented following severe floods (Carline & McCullough, 2003; Roghair, Dolloff & Underwood, 2002; Warren, Ernst & Baldigo, 2009; Milner et al., 2012). Direct effects involve displacement-related mortality and destruction of incubating eggs while indirect effects relate to habitat changes which can affect carrying capacity or disproportionately favor one species or guild over others (Elwood & Waters, 1969). A better understanding of the short- and long-term impacts of floods on fish assemblages and their recovery is needed to identify factors which afford resistance and/or resilience to stream biota and entire ecosystems. Since most climate change models predict increased frequency of extreme hydrologic events over the next several decades (Rosenzweig et al., 2011), such information will become invaluable in order to sustain biodiversity of native fish communities and protect species of concern.

A number of factors contribute to the overall effect of floods on lotic fish communities including flood magnitude, the availability and accessibility 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 (Strange, Moyle & Foin, 1993; Fausch et al., 2001; Harvey, 1987). Bed mobilizing events that occur while eggs are in the gravel or elevated flows shortly after
fry have emerged are particularly damaging (Warren, Ernst & Baldigo, 2009). In salmonids, the risk of displacement during high flows is greatest when fry emerge from the gravel and enter the free-feeding stage where they can be displaced by velocities as low as 0.1 m/s (Heggenes & Traen, 1988). This life stage is known as the critical period because the size of the 0+ year class (herein termed young-of-the-year or YOY) determines the initial cohort size and limits the future strength of this year class through time (Nislow, Einum & Folt, 2004). Thus, it is not surprising that the strength of salmonid year classes has often been correlated to 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 fall spawning salmonids, such as brown trout (Salmo trutta) and brook trout (Salvelinus fontinalis), and potentially provide a competitive advantage to spring spawners, such as rainbow trout (Oncorhynchus mykiss) (Warren, Ernst & Baldigo, 2009; Strange, Moyle & Foin, 1993). Conversely, late spring floods that occur after the fry of fall spawners have emerged and advanced beyond the critical period have been shown to scour the eggs of spring spawners or cause mortality to their newly emerged fry, thereby favoring fall spawners (Seegrist & Gard, 1972). Relatively few studies, however, have documented the effects of summer floods on fish assemblages (Bischoff & Wolter, 2001; Jurajda, Reichard & Smith, 2006), or sympatric trout species (Pearsons, Li & Lamberti, 1992; Nislow et al., 2002). In the northeastern United States, large summer floods have generally occurred less frequently than spring, snow-melt driven floods which suggests that resident fish species may be less well
adapted and more susceptible to summer floods and thus their impact could be more severe (Giller, 1996; Jones & Petreman, 2013).

In 2011, a late summer hurricane caused catastrophic flooding in streams throughout the eastern United States which provided an opportunity to quantify the effects on stream fish communities in a mountainous watershed. Fish communities were previously surveyed annually at 18 main stem and tributary sites within the Upper Esopus Creek (hereafter termed the Esopus) Basin in the Catskill Mountains of southeastern New York from 2009-2011 as part of a study to evaluate the effects of turbidity and supplemental flows from an inter-basin aqueduct (Shandaken Tunnel) on local fish assemblages (Figure 1).

**Figure 1** - Map showing the locations of study sites on the Upper Esopus Creek and tributaries and the years in which they were surveyed, 2009-2013.
Approximately one week after the conclusion of the summer 2011 surveys, the Catskill Mountain Region experienced severe flooding from Tropical Storm Irene on August 28, 2011. Rainfall at Slide Mountain in the headwaters of the Esopus totaled 29.3 cm during the passage of Irene (Lumia, Firda & Smith, 2014). The annual exceedance probabilities (AEP) for peak flows at five permanent U.S. Geological Survey (USGS) streamgages in this watershed ranged from 0.143 to 0.008, which corresponded to flood recurrence intervals (inverse of AEP) of between 7 and >100 years (Table I).

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

<table>
<thead>
<tr>
<th>Stream and site name</th>
<th>Site ID</th>
<th>DA (km$^2$)</th>
<th>Elevation (m)</th>
<th>Peak discharge (m$^3$ s$^{-1}$)</th>
<th>Annual exceedance probability</th>
<th>Recurrence interval (yrs)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fox Hollow</td>
<td>fox</td>
<td>10.3</td>
<td>309.4</td>
<td>-</td>
<td>0.067$^1$</td>
<td>15$^1$</td>
</tr>
<tr>
<td>Broadstreet Hollow</td>
<td>broad</td>
<td>23.7</td>
<td>295.8</td>
<td>-</td>
<td>0.015$^2$</td>
<td>65$^2$</td>
</tr>
<tr>
<td>Birch Creek</td>
<td>birch</td>
<td>32.4</td>
<td>377.4</td>
<td>41.3</td>
<td>0.143</td>
<td>7</td>
</tr>
<tr>
<td>Woodland Valley Creek</td>
<td>wood</td>
<td>53.4</td>
<td>267.6</td>
<td>189.4</td>
<td>0.067</td>
<td>15</td>
</tr>
<tr>
<td>Stony Clove Creek</td>
<td>stoc1</td>
<td>83.9</td>
<td>245.2</td>
<td>404.9$^3$</td>
<td>0.012$^3$</td>
<td>80$^3$</td>
</tr>
<tr>
<td>Esopus Creek at Oliverea</td>
<td>esop0</td>
<td>30.3</td>
<td>454.5</td>
<td>-</td>
<td>0.008$^4$</td>
<td>&gt;100$^4$</td>
</tr>
<tr>
<td>Esopus Creek at Big Indian</td>
<td>esop2</td>
<td>111.9</td>
<td>354.9</td>
<td>-</td>
<td>0.008$^4$</td>
<td>&gt;100$^4$</td>
</tr>
<tr>
<td>Esopus Creek at Allaben</td>
<td>esop3a</td>
<td>165.0</td>
<td>304.6</td>
<td>829.7</td>
<td>0.008</td>
<td>&gt;100</td>
</tr>
<tr>
<td>Esopus Creek at Boiceville</td>
<td>esop6</td>
<td>497.3</td>
<td>188.8</td>
<td>2146.4</td>
<td>0.014</td>
<td>70</td>
</tr>
</tbody>
</table>

$^1$ Ungaged site; AEP and recurrence interval taken from USGS streamgage at wood
$^2$ Ungaged site; AEP and recurrence interval taken as averages from USGS streamgages on adjacent streams (stoc1 and Bushnellsville Creek)
$^3$ Data from a USGS streamgage located on Stony Clove Creek 3 km upstream of stoc1
$^4$ Ungaged site; AEP and recurrence interval taken from USGS streamgage at esop3a
Discharge had nearly receded to pre-flood levels by September 7, 2011, when another flood from the remnants of Tropical Storm Lee affected the watershed 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 USGS streamgages. In-stream and riparian habitats were severely altered at most study sites and emergency repair efforts by local municipalities and private landowners further affected several study reaches. The most frequently observed habitat alterations were loss of riparian vegetation, loss of canopy (greater sunlight exposure), and decreased channel complexity (Figure 2).

![Figure 2](image)

**Figure 2** - The confluence of the Upper Esopus Creek and the Shandaken Tunnel in August, 2009 prior to Irene (left) and in September, 2011 less than one month after Irene (right).

Post-flood surveys of fish assemblages were completed at nine of the 18 previously surveyed sites during 2012 and 2013 to increase our understanding of the response and resilience of fish assemblages to extreme hydrologic events. Specific hypotheses tested were that: (1) fish community metrics would show a basin-wide decline, (2) fish community composition would be altered, and (3) young-of-the-year trout would be disproportionately affected compared to other life stages. The effects of
interspecific competition and different life history strategies on the severity of impact and degree of recovery in both spring and fall spawning trout populations were also explored.

*Study Area*

The Upper Esopus Creek is located in the south central Catskill Mountain Region of southeastern New York, USA (Figure 1). The Esopus follows a 41.8 km semi-circular, clockwise course from its headwaters at Winnisook Lake, around Panther Mountain, to its terminus at the Ashokan Reservoir downstream of Boiceville, NY. The 497.3 km² watershed is over 95% forested and its surficial geology features 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 (Figure 2), a 29-km aqueduct connecting the reservoir to the Esopus near Shandakan, NY, approximately 23 km downstream from its source. 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 main stem of the Esopus and five were located on major tributaries near their confluence with the Esopus (Table I).

*Methods*

*Hydrology*

Peak discharge, AEP, and recurrence intervals for five study sites located at or near active USGS streamgages were provided by Lumia, Firda & Smith (2014). At each of the four ungaged sites, the AEP was assumed to equal that of a gage(s) on an adjacent reach or nearby stream of similar orientation. We did not estimate peak discharge for the ungaged study sites (Table I).
Fish Surveys

Two similar, but slightly different sampling techniques were used to quantify fish assemblages at small- and large-channel study sites (reaches) between late-June and mid-August each year from 2009-2013. These reaches ranged from 10 to 20 mean channel widths long and typically encompassed one or two complete geomorphic channel-unit sequences (Fitzpatrick *et al*., 1998; Simonson, Lyons & Kanehl, 1994; Meador, McIntyre & Pollack, 2003). Fish were collected with a backpack electrofisher and three to five personnel to net fish using a three-pass depletion method. At sites with narrow channels (width <15 m) blocking seine nets were placed completely across the channel at the upstream and downstream end of study reaches. At sites with channels greater than 15 m wide, three (replicate) surveys were conducted in relatively small near-shore sub-reaches. At each sub-reach, one blocking seine was affixed to the bank and then stretched perpendicular to the bank and attached to a rock or rebar (6 to 8 m from shore); a second 25-m seine was oriented downstream and parallel to shore and then attached to a second rock or rebar (also 6 to 8 m from shore); and a third seine was placed between the second rebar and shore. Fish were collected and all individuals were processed separately for each of the three successive electrofishing passes. All fish were identified to species and the lengths and weights for individuals longer than 150 mm were recorded. The lengths and weights for small abundant species (e.g., some minnows) were obtained from at least 30 individuals; after which pooled weights (and counts) were recorded by species in batches of up to 25 fish. All fish were subsequently returned to the stream.

Within each study reach and sub-reach, 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 center, 25%, and 75% of each cross section) along each of the 10 transects. The total length and mean width were used to calculate sample 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 (Van Deventer & Platts, 1985; Zippin, 1958; Van Deventer & Platts, 1983). These values were divided by the total area sampled at each study site to estimate density (number) and biomass (grams) of fish, in the local community or species’ population per unit area. Two components of ecosystem diversity; breadth or size (total species richness) and heterogeneity (Simpson’s diversity index) were also generated for each site (Simpson, 1949; Whittaker, 1975). Simpson’s diversity index (reinterpreted as 1-D) uses the number of species present and the relative abundance of each species to calculate a metric that ranges from 0 to 1; zero indicates no diversity (e.g. a single dominant species) and higher values indicate multiple species with similar relative abundances. 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 group levels. Most captured fish were grouped by family as Salmonidae (*Salmo trutta, Oncorhynchus mykiss, Salvelinus fontinalis*), Cottidae (*Cottus cognatus*), Cyprinidae (*Rhinichthys atratulus, Rhinichthys cataractae, Pimephales promelas, Exoglossum maxilliguinga, Semotilus corporalis, Notemigonus crysoleucas, Pimephales notatus, *Luxilus cornutus, and Semotilus atromaculatus*), or Catostomidae (*Catostomus commersonii and Catostomus catostomus*) (hereafter referred to by the common names trout, sculpin, minnow, and sucker) and the remaining 11 uncommon species were grouped as “other”. Thus, annual variations in density and biomass could be evaluated for each group at individual sites and across sites. Additional analysis of spatial patterns in fish community composition was conducted using non-metric multidimensional scaling (MDS) of square-root transformed species density data (Kruskal, 1964; Shepard, 1962) using Primer-E v6 software (Clarke & Warwick, 2001b). The MDS ordination generates an arrangement of samples in ‘species space’ according to the non-parametric ranks of their Bray–Curtis similarities (Clarke & Warwick, 2001a). An Analysis of Similarities (ANOSIM) was also used to test the second hypothesis that species assemblages differed between pre-flood and post-flood surveys.

Changes in the dynamics of 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 because they were only encountered sporadically and always at low densities. 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 for YOY and older individuals for each species and year.

We evaluated the effect of flood magnitude on fish communities using Spearman correlation between changes in fish metrics and landscape/hydrologic characteristics. Landscape/hydrologic variables for each site were drainage area, elevation, and AEP. Fish metrics were change (pre-flood versus post-flood) in: community density, community biomass, brown trout density, brown trout biomass, rainbow trout density and rainbow trout biomass. Change in fish metrics was calculated two different ways: 1) the post-flood (2012) condition minus the mean pre-flood (2009, 2010, and 2011) condition and 2) the 2012 condition minus the 2011 condition.

Results

Hydrology

The magnitude of flooding varied widely across study sites in the Esopus Basin. AEP ranged from 0.143 at birch to 0.008 at esop0, esop2, and esop3a (Table I). The most severe floods generally occurred at main stem Esopus sites (0.014-0.008 AEP) while floods were more variable in tributaries, ranging from 0.143-0.012 AEP. Peak discharge ranged from 41.3 m$^3$ s$^{-1}$ at birch to 2146.4 m$^3$ s$^{-1}$ at esop6 and generally was consistent with differences in drainage area.
Fish Community Metrics

Annual comparisons of mean community metrics across sites showed that each metric increased between the pre-flood and post-flood year and that density and biomass were the most variable (Figure 3). Mean species richness ($P = 0.422$, rANOVA) and mean species diversity ($P = 0.299$) did not differ significantly between years. Mean community density was 1900 fish 0.1ha$^{-1}$ in 2009, 1312 fish 0.1ha$^{-1}$ in 2010, 753 fish 0.1ha$^{-1}$ in 2011, 1498 fish 0.1ha$^{-1}$ in 2012, and 1418 fish 0.1ha$^{-1}$ in 2013 and differed significantly between years ($P = 0.006$). Pairwise comparisons showed mean density was significantly lower in 2011 than in 2009 ($P < 0.001$, Fisher’s LSD test), 2012 ($P = 0.012$), and 2013 ($P = 0.024$) and that mean density also differed between 2009 and 2010 ($P = 0.044$). Mean community biomass was 13022 g 0.1ha$^{-1}$ in 2009, 8851 g 0.1ha$^{-1}$ in 2010, 5060 g 0.1ha$^{-1}$ in 2011, 10464 g 0.1ha$^{-1}$ in 2012, and 9793 g 0.1ha$^{-1}$ in 2013 and differed significantly between years ($P = 0.042$). Pairwise comparisons showed that mean biomass in 2011 differed significantly from that in 2009 ($P = 0.003$) and 2012 ($P = 0.035$). None of the four metrics differed significantly ($P > 0.05$, paired t-test) between the pooled pre-flood (2009-2011) and post-flood (2012-2013) periods.
Year to year differences in species richness and diversity at individual sites (Appendix I) were consistently small and suggest that there were only minor or no significant effects of the flood on these metrics; thus these findings are not described in more detail herein. In contrast, the annual differences in total density and biomass of fish communities at individual study sites (Figure 4) were frequently large which was consistent with the pooled analysis (above). Total density was generally high and peaked at five of the nine sites during 2009, decreased at eight sites during 2010, and decreased further at all nine sites during 2011 reaching lowest levels at six sites. However, total fish density increased at eight of the nine sites (esop0 experienced a 2% decrease) during the
first post-flood year (2012), and did not change markedly at most sites between 2012 and 2013 (Figure 4a). Total density peaked at four of the nine sites (fox, broad, birch, and stoc1) peaked during the post-flood years. Similar to density, total community biomass was generally at or near its highest levels at most sites during 2009-10, decreased at seven sites between 2010 and 2011 reaching lowest levels at six sites, and increased to relatively high levels again during 2012 and 2013. Total community biomass at four study sites peaked during either 2012 or 2013 (Figure 4b).

Figure 4 - Stacked bar graphs showing (a) density and (b) biomass of fish communities at each site for each year. Sites are arranged by drainage area and whiskers show the 95% confidence intervals based on depletion population estimates for each year.
Fish Community Structure

The structure of fish communities was evaluated using (a) changes in the density and biomass of the five major species’ groups and (b) an MDS ordination showing similarity between fish assemblages by site and year. The three groups with the greatest mean density were minnow, sculpin, and trout, respectively, and this order was maintained during each year of the study (Figure 5a). Mean density of these three dominant groups declined together from 2009-2011 but then increased in 2012, and diverged in 2013. The mean biomass of trout was consistently higher than that of other groups (Figure 5b) while the mean biomass of minnow and sculpin was typically 50-to-75% of trout biomass during most years. The mean biomass of sucker was relatively large (unlike sucker density) and highly variable among years. Mean biomass for these four dominant groups declined from 2009 to 2010 and again from 2010 to 2011; increased substantially during 2012; and either changed negligibly (minnow and sculpin), increased (trout), or decreased (sucker) during 2013. Except for sucker, the year-to-year changes in biomass for all groups generally reflected comparable changes in group density. The high sucker biomass noted during 2009 and 2012 and corresponding low densities resulted from the collection of a few large individuals at one or two sites during both years.
Figure 5 - Stacked bar graphs showing mean (n = 9) (a) density and (b) biomass for each species’ group during each year.

The wide dispersion of sites across both axes of the MDS ordination (Figure 6) indicates fish assemblages differed considerably between sites, but not between pre-flood (2009-11) and post-flood (2012-13) surveys. ANOSIM confirmed that there were 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 sampled during 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, main stem sites with the largest drainage areas and high species richness, such as esop3a and esop6, generally grouped to the right.
Figure 6 - Non-metric multidimensional scaling ordination showing fish assemblages at each site during each year.

Trout Populations

Total density and biomass of the brown trout population (basin-wide) generally followed the pattern exhibited by the overall fish community. Density of brown trout over the first three years decreased from 190 fish 0.1ha⁻¹ in 2009 to 90 fish 0.1ha⁻¹ in 2010 to 23 fish 0.1ha⁻¹ in 2011. Following the summer 2011 flood, density of brown trout peaked at 246 fish 0.1ha⁻¹ in 2012 and declined to 104 fish 0.1ha⁻¹ in 2013 (Table II). Similarly, Biomass of brown trout decreased from 3950 g 0.1ha⁻¹ in 2009 to 2752 g 0.1ha⁻¹ in 2010 to 1161 g 0.1ha⁻¹ in 2011 before increasing to 2254 g 0.1ha⁻¹ in 2012 and peaking at 3985 g 0.1ha⁻¹ in 2013. These annual variations in the density and biomass of brown trout produced the greatest mean weight (48.2 g) during 2011, the lowest mean weight (9.2 g) during 2012, and intermediate values for all other years (Table II).
Annual variations in the density and biomass of the basin-wide rainbow trout population differed considerably from those observed for brown trout. Mean density of rainbow trout was relatively constant during 2009 (114 fish 0.1ha⁻¹) and 2010 (118 fish 0.1ha⁻¹), declined in 2011 (38 fish 0.1ha⁻¹), remained low after the flood in 2012 (43 fish 0.1ha⁻¹), and declined to its lowest level (16 fish 0.1ha⁻¹) in 2013 (Table II). Biomass of rainbow trout increased steadily from 491 g 0.1ha⁻¹ to 1139 g 0.1ha⁻¹ between 2009 and 2012 and then declined to 481 g 0.1ha⁻¹ in 2013. Unlike brown trout, the decreases in density and the increases in biomass of rainbow trout produced yearly increases in mean weight of individual fish (Table II).

*Table II* – Mean (n = 9) density and biomass estimates from 3-pass depletion methods, percentage young-of-the-year (YOY) and mean weight from raw data of all sampled trout in each year irrespective of site, and YOY density and non-YOY density derived from estimated populations densities and percentage YOY of brown trout and rainbow trout.

<table>
<thead>
<tr>
<th>Year</th>
<th>Density of population (fish 0.1ha⁻¹)</th>
<th>Percent YOY (%)</th>
<th>Density of YOY (fish 0.1ha⁻¹)</th>
<th>Density of non-YOY (fish 0.1ha⁻¹)</th>
<th>Biomass of population (g 0.1ha⁻¹)</th>
<th>Mean weight (g)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Brown Trout</td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>2009</td>
<td>190</td>
<td>73</td>
<td>138</td>
<td>52</td>
<td>3950</td>
<td>20.7</td>
</tr>
<tr>
<td>2010</td>
<td>90</td>
<td>39</td>
<td>35</td>
<td>55</td>
<td>2752</td>
<td>36.8</td>
</tr>
<tr>
<td>2011</td>
<td>23</td>
<td>49</td>
<td>11</td>
<td>12</td>
<td>1161</td>
<td>48.2</td>
</tr>
<tr>
<td>2012</td>
<td>246</td>
<td>94</td>
<td>231</td>
<td>15</td>
<td>2254</td>
<td>9.2</td>
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<tr>
<td>2013</td>
<td>104</td>
<td>51</td>
<td>53</td>
<td>51</td>
<td>3985</td>
<td>36.6</td>
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<td>Rainbow trout</td>
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<tr>
<td>2009</td>
<td>114</td>
<td>89.8</td>
<td>102</td>
<td>12</td>
<td>491</td>
<td>5.1</td>
</tr>
<tr>
<td>2010</td>
<td>118</td>
<td>80.4</td>
<td>95</td>
<td>23</td>
<td>590</td>
<td>5.8</td>
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<tr>
<td>2011</td>
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<td>56.6</td>
<td>21</td>
<td>17</td>
<td>876</td>
<td>22.8</td>
</tr>
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<td>21</td>
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<td>26.4</td>
</tr>
<tr>
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<td>16</td>
<td>52.5</td>
<td>8</td>
<td>8</td>
<td>481</td>
<td>29.8</td>
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</table>
The size of various age classes for basin-wide brown trout and rainbow trout populations varied considerably between years and differed from each other within the same years (Figure 7). Young-of-the-year and yearling (age 1+) brown trout were abundant during 2009, decreased in 2010, and declined to their lowest abundance during 2011. The estimated density of YOY brown trout was lowest (11 fish 0.1ha\(^{-1}\)) immediately before the flood (2011) and highest (231 fish 0.1ha\(^{-1}\)) the first year after the flood (Table II). Marked increases in YOY brown trout were observed between 2011 and 2012 at most sites. Most notably, YOY brown trout were absent at fox during 2011, yet an estimated 1286 fish 0.1ha\(^{-1}\) were present during the 2012 survey (Appendix I). During the following year (2013), the numbers of brown trout YOY decreased and the numbers of yearlings increased which reflected a relatively well-balanced age structure, not seen since 2009 (Figure 7). The decrease in density of rainbow trout (and change in age structure) was similar to that observed for brown trout from 2009 to 2011, but the population did not recover in 2012 and 2013 (Figure 7, Table II). Density of YOY rainbow trout declined from 102 fish 0.1ha\(^{-1}\) in 2009 to 8 fish 0.1ha\(^{-1}\) in 2013 (Table II). Only fox and esop3a had relatively large increases in YOY rainbow trout between 2011 and 2012 as seen with brown trout at many sites (Appendix I).
Figure 7 - Length frequency distributions showing the lengths of all brown trout and rainbow trout captured from nine study sites during 2009-2013.
The percentage of YOY brown trout was highest (94%) the first year after the flood (2012) and varied between 39 and 73 percent during all other years (Table II). The percentage of YOY rainbow trout was highest (90%) during 2009 and decreased from 80% during 2010 to 51-57% during 2011-2013. Recruitment of YOY brown trout appeared to increase greatly after the flood, whereas, recruitment of rainbow trout declined to five-year lows (near 52%) during both years after the flood.

**Relationships between Landscape/Hydrologic Variables and Fish Response**

Major landscape and hydrologic variables were not strongly correlated with fish community metrics. There were no significant correlations with landscape/hydrologic variables when changes in fish metrics for each site were calculated as the average of the pre-flood surveys subtracted from the 2012 surveys. A similar comparison using only 2011 versus 2012 data found one significant correlation ($r = -0.667, P = 0.043$) between drainage area and change in brown trout density (Table III). Smaller drainage area was associated with larger increases in brown trout density while larger drainage area was associated with smaller increases in brown trout density. Weaker correlations between AEP and change in community density ($r = 0.630, P = 0.0583$) and AEP and change in brown trout density ($r = 0.570, P = 0.0988$) were also evident but were not significant.
Table III - Spearman correlations ($r$) and $P$-values between change in fish metrics (2012-2011) with elevation, drainage area, and annual exceedance probability (AEP) for all nine sites ($n = 9$). Asterisk denotes a significant ($P < 0.05$) correlation.

<table>
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<tr>
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<th>Elevation $r$</th>
<th>Drainage Area $r$</th>
<th>AEP $r$</th>
</tr>
</thead>
<tbody>
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<td>Community Density</td>
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<td>-0.550</td>
<td>0.630</td>
</tr>
<tr>
<td>Community Biomass</td>
<td>0.333</td>
<td>-0.283</td>
<td>0.375</td>
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<tr>
<td>Brown Trout Density</td>
<td>0.100</td>
<td>-0.667*</td>
<td>0.570</td>
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<tr>
<td>Brown Trout Biomass</td>
<td>0.233</td>
<td>-0.050</td>
<td>0.119</td>
</tr>
<tr>
<td>Rainbow Trout Density</td>
<td>0.350</td>
<td>-0.133</td>
<td>0.281</td>
</tr>
<tr>
<td>Rainbow Trout Biomass</td>
<td>0.433</td>
<td>-0.017</td>
<td>0.187</td>
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</tbody>
</table>

Discussion

The factors that control the short-term response of fish assemblages to extreme floods in the Esopus are apparently complex. The magnitude of the flood (Resh et al., 1988) and the availability of refuges (e.g., floodplains and backwaters) (Jowett & Richardson, 1989; Pearsons, Li & Lamberti, 1992; Lake, 2000), are two key factors that should control the severity of impacts. Although the flood magnitude varied widely across sites in the Esopus Basin, peak discharges at many sites exceeded the 0.0167 AEP discharge (Table I); comparable or greater to that observed in many other studies that detected severe fishery impacts (Carline & McCullough, 2003; Jowett & Richardson, 1989). Most correlations between flood magnitude (AEP) and changes in fish metrics in the Esopus, however, identified no significant relationships. In addition, channels for almost all study sites were highly incised, or adjacent to roads, and essentially disconnected from their flood plain before the flood event. This is concerning because the flood plain is normally the primary refuge area for fish during major floods (Schwartz & Herricks, 2005; Ross & Baker, 1983). Despite this, fish communities of the Esopus were
relatively unaffected at the basin scale. In fact, 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 and the weak correlations between flood size and several fish metrics were surprising given the flood magnitudes and poor floodplain connectivity. Apparently additional factors, acting directly and (or) indirectly, increased the effective resistance or resilience of local fish assemblages to the extreme flood caused by Irene in the Upper Esopus Basin.

The timing of the late summer flood triggered by Irene may help explain why more severe impacts to the fish community were not observed. There is increasing evidence that the timing of hydrologic events can strongly affect the severity and nature of ecological impacts (Fausch et al., 2001; Harvey, 1987; Pearsons, Li & Lamberti, 1992; Giller, 1996). Late summer flooding could have had comparatively minor community impacts in the Esopus for several reasons. 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 approximately 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 (Figure 7) and can hold their positions against much greater velocities (Heggenes & Traaen, 1988). Although much smaller than age 0+ trout, the YOY of other abundant species such as slimy sculpin (Cottus cognatus) and most cyprinids would have also advanced beyond the critical period by late August (Harvey, 1987). Second, as poikilothermic animals, fish metabolism is higher during warm periods, which suggests
they may be able to withstand greater water velocities, and thus, resist displacement (caused by floods) more effectively than during cold months (Heggenes & Traaen, 1988; Glova & McInerney, 1977). Accordingly, more severe impacts would be expected from a flood of similar magnitude occurring during winter or spring.

Although the summer flood did not adversely affect the Esopus fish community basin-wide, analogous responses cannot always be assumed or generalized for individual sites. The effects that the 2011 flood had on fish assemblages varied considerably among study sites. For example, except for esop0, increases in total density following the flood were greater at sites with small drainage areas (mean increase of 192% at fox, broad, birch, wood, and stoc1) and comparatively smaller at sites with large drainage areas (mean increase of 37% at esop2, esop3a, and esop6). In addition, total biomass increased by a mean 158% at eight sites while it decreased by 62% at the largest drainage area site (esop6) between 2011 and 2012. Although hydrologic data indicate some of the sites with small drainage areas experienced floods of lesser magnitude during Irene, drainage area was not significantly correlated with the degree of change in fish metrics. The effects of emergency channel repairs at two study sites immediately following the flood was not assessed, but may be another potentially complicating factor. Emergency repairs occurred at stoc1 and esop3a and resulted in relatively homogeneous stream channels. Although adverse ecological effects from such actions might be expected, total density and biomass at these sites actually increased between 2011 and 2012, and again between 2012 and 2013. This suggests that fish assemblages were either minimally affected by the additional anthropogenic disturbance or that they had sufficient time to recolonize disturbed reaches. The fact that fish assemblages at two sites (esop0 and esop6) may have
been adversely affected by the flood suggests that fish communities at all study sites did not respond to the flood in unison.

The unusually poor condition of the Esopus fish communities during 2011 was surprising and complicated the analysis of flood impacts and recovery. Most basin-wide analyses, site-specific metrics, and community composition analyses indicate that density and biomass of fish communities differed most or were at their lowest levels during 2011. A combination of adverse hydrologic and thermal conditions during 2010 and 2011 may be responsible for this observation. A prolonged period of low flows occurred during the summer of 2010. Mean daily discharge at esop6 for the months June-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 more from historical averages; for example, mean daily discharge at wood (0.5 m$^3$ s$^{-1}$) during summer 2010 was only 36% of the long-term average (1.4 m$^3$ s$^{-1}$) (USGS, 2011b). These extremely low flows coincided with an unusually long period (July 16-August 20, 2010) of warm stream temperatures (mean 22 ºC at esop6) (USGS, 2011a) (Figure 8).
Reduced habitat volume and surface area caused by low flows may have further affected many fish species in the Esopus during summer 2010. Decreases in the quantity and quality of riverine habitat due to drought have been shown to produce mortality in fish and macroinvertebrates with a disproportionate effect on the organisms that depend most on moving water (Bond, Lake & Arthington, 2008; Lake, 2000; Magoulick & Kobza, 2003). The 2010 fish surveys in the Esopus were conducted between late June and early August, which may have been too early to detect the full effects of the 2010 drought. Therefore, any effects that the 2010 drought had on fish assemblages would have been most evident during the 2011 surveys, and could explain why metrics were atypical during 2011.

The large increase in the number of YOY brown trout during 2012 suggests a rapid recovery and indicates the species was highly resilient to the extreme summer flood. The specific causes for the observed increase were not investigated directly, however, the life history strategy of this species provides the basis for several inferences.
First, post-flood changes within stream channels may have exposed more trout spawning habitat (gravel), or increased the quality of existing spawning habitat. Major floods have been shown to flush detrimental fine sediment from streams and generally coarsen the substrate (Herbst & Cooper, 2010; Carlile & McCullah, 2003; Jowett & Richardson, 1989; Ortlepp & Mürle, 2003). This 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 spawning beds or beds of higher quality, thereby increasing survival of overwintering trout eggs during winter 2011-2012 and swim-up fry during spring 2012. Second, stream flows during winter 2011-2012 and spring 2012 were moderate and relatively stable; ideal for incubating embryos and emerging fry (Lobon-Cervia, 2004) and could have led to unusually high brown trout recruitment. Third, the low density of all species observed in summer 2011 suggests that upon emergence, the 2012 year class had fewer large predators to evade and fewer small competitors to contest available food and space. Fourth, all study sites in this basin are located within 2.6-to-35.4 km of the 34 km² Ashokan Reservoir (Figure 1). Mature brown trout from the Ashokan Reservoir spawn in the Esopus and its tributaries every fall. 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 many resident spawners had been eliminated from some reaches by fall 2011. Together, these factors could have effectively increased brown trout recruitment to the 2012 year class. Though brown trout spawning strategies appeared to increase the resilience of their populations to the summer 2011 flood, the opposite effect would be expected if comparable floods occurred between November and April.
The divergent responses of brown trout and rainbow trout populations to the summer flood could have its basis in competitive interactions. Unlike brown trout which exhibited a marked increase in YOY following the flood, the density of YOY rainbow trout was essentially unchanged between 2011 and 2012 (Table II). Increases in the density of brown trout following the flood and the resulting increased interspecific competition might be responsible for the low densities of rainbow trout during 2012 and 2013. Brown trout are fall spawners, whereas rainbow trout are spring spawners, thus, YOY brown trout emerge earlier, are larger, and may potentially outcompete YOY rainbows during the summer months (Gatz Jr, Sale & Loar, 1987; Strange, Moyle & Foin, 1993). Conversely, the fall spawning life history of brown trout increases the risk of egg loss if fall and winter floods occur. Several investigations have documented the regulating effect of winter floods on the dominance of fall versus spring spawning salmonids (Strange, Moyle & Foin, 1993; Seegrist & Gard, 1972); specifically that YOY brown or brook trout were generally more numerous than YOY rainbow trout except during years when floods scoured the eggs of the fall spawners. In fact, Warren, Ernst & Baldigo (2009) recently observed this phenomenon on two streams in the Upper Esopus Basin. Thus, the summer 2011 flood and factors unrelated to this flood (favorable flows during winter 2011-2012 and reduced predation or competition) may have positively affected recruitment and the size of the brown trout year class during 2012. Consequently, this large brown trout cohort may be partially responsible for poor rainbow trout recruitment (and size of year classes) during 2012 and 2013. The poor rainbow trout year class during 2011, on the other hand, cannot be easily explained by interspecific competition and may relate to stochastic factors (e.g., drought in 2010 and
smaller floods in 2010-2011) that depressed the entire fish community; suggesting that both biotic and abiotic mechanisms regulate fish populations in this system.

Several large floods, besides those associated with Irene, also affected the basin during the 2011 water year (October 1st, 2010 – September 30th, 2011) and could have disproportionately affected survival and recruitment of YOY trout during 2011 and 2012. The second, third, and fifth largest floods observed at esop6 during the five year study period took place on October 1, 2010, December 1, 2010, and March 11, 2011, respectively (Figure 9).

![Figure 9: Line graph showing unit (15-minute) values for discharge (m³ s⁻¹) at esop6 from 10/1/2008-10/1/2013.](image)

Similar to Irene, the October 1, 2010 flood likely had minimal impacts on trout populations because it did not coincide with spawning, embryo incubation, or the critical period of fry emergence. In contrast, although peak flow during the December 1, 2010 event was only about half of that caused by Irene (1127 m³ s⁻¹ at esop6), this bed-mobilizing event likely displaced and substantially reduced the survival of overwintering
brown trout eggs. In addition, the March 2011 flood \((773 \text{ m}^3 \text{ s}^{-1} \text{ at esop6})\) likely further limited the size of that year class but should have occurred prior to rainbow trout spawning \((\text{Smith, 1985})\). Given strong interspecific competition between the two trout species, the December and March floods should have increased the densities of the 2011 rainbow trout year class relative to brown trout \((\text{Warren, Ernst & Baldigo, 2009; Seegrist & Gard, 1972})\). Instead, the density of YOY rainbow trout decreased from 95 fish \(0.1\text{ha}^{-1}\) during 2010 to 21 fish \(0.1\text{ha}^{-1}\) during 2011, and YOY brown trout decreased from 35 fish \(0.1\text{ha}^{-1}\) during 2010 to 11 fish \(0.1\text{ha}^{-1}\) during 2011 \((\text{Table II})\). The very poor 2011 year classes for both trout species suggest that either, (a) the 2010 drought had a comparably larger adverse effect on both trout species, or (b) rainbow trout spawning occurred earlier than expected and was disrupted by the March 2011 flood.

The timing of extreme hydrologic events and various life history strategies of different species evidently interact to produce a range of responses to both floods and droughts \((\text{Fausch et al., 2001; Strange, Moyle & Foin, 1993; Jones & Petreman, 2013})\). The low numbers of yearling trout during 2012 suggest that the flooding from Irene may have been sufficiently large to eradicate YOY trout from most study sites, yet very few YOY trout of any species were residing at any site before the flood occurred \((\text{with the exception of rainbow trout at broad in 2011})\). The low density of YOY brown trout encountered during the 2011 surveys indicates that other factors adversely affected the strength of that year class before the August 2011 flood occurred. Discharge across the basin was at drought levels during the summer of 2010 and maximum stream temperatures at esop6 reached 26.9 °C \((\text{USGS, 2011a})\) – near the 1-day maximum temperature (MAXT) tolerance for brown trout of 27.6 °C as defined by Wehrly, Wang &
Mitro (2007). Additionally, the average temperature observed at esop6 from July 16-August 20, 2010 (22 ºC), exceeded the 28-day mean thermal tolerance limit (MEANT) for brown trout (Wehrly, Wang & Mitro, 2007). This suggests cold-water fishes that could not find thermal refuge had decreased physiological condition and possibly increased mortality. A recent study (Ross, 2012) confirmed that brown trout in the Upper Esopus Creek showed signs of secondary-level stress response and were generally in poor condition in 2010. Consequently, the number and condition of mature spawning trout (in the fall of 2010 and spring of 2011) may have been unusually low. These circumstances, combined with likely bed disturbance, egg loss, and fry displacement from the fall and winter floods, likely depressed the size of the 2011 year class of both trout species. As a consequence, the changes in trout (and community) 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 flood and recovery from the 2010 drought and winter 2011 floods.

Conclusions

The findings of this study have a number of implications for assessing and managing natural resources (fishery, recreation, and water use) in the Esopus and other streams of the region. First and most notably, they underscore the value of long-term monitoring programs. The analyses provided herein were possible only because three consecutive years of baseline fish community data were available prior to the flood. The preexisting fish community metrics placed the post-flood results into the proper 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 datasets, not only locally, but also regionally, because precipitation, temperature, and stream discharge are expected to increase in the Catskill Region (Rosenzweig et al., 2011) and northeastern US (Fowler & Hennessy, 1995; Jentsch, Kreyling & Beierkuhnlein, 2007) in the future. Lastly, existing trends in densities of two trout species suggest that habitat conditions in some parts of the basin may be shifting to favor brown trout over rainbow trout. Additional data collection and analysis, however, are needed to determine whether this is a normal cyclical fluctuation in population densities, related to some other biotic or abiotic stressor in the system, or a long-term phenomenon related to climate change. The answers to these questions are important because they have broad geographic relevance and affect how both trout species are managed by the New York State Department of Environmental Conservation.

Despite atypical fish assemblages at many sites in the Upper Esopus Creek Basin during summer 2011, three years of pre-flood data enabled us to test and reject two of the three proposed hypotheses. First, our results show that fish community metrics were not adversely affected basin-wide 10-11 months after the flood. Second, the composition of fish communities after the flood did not change significantly 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 that the flood had 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 between 2011 and 2012. Additionally, the high density of YOY brown trout one year after the flood is one of our most significant results
and suggests excellent recruitment immediately following the flood. Together, these findings demonstrate that resident fish species and their communities in the Upper Esopus Creek Basin were relatively resistant to effects of the summer 2011 flood caused by Irene. Although additional study will be needed to validate these conclusions, the effects of drought and (or) floods that coincide with critical life stages may constitute a greater disturbance to lotic fish communities than a severe late-summer flood.
References


35


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

<table>
<thead>
<tr>
<th>Site ID</th>
<th>Community Density (fish 0.1ha⁻¹)</th>
<th>Community Biomass (g 0.1ha⁻¹)</th>
<th>Community Richness</th>
<th>Community Diversity (1-D)</th>
<th>Brown Trout Density (fish 0.1ha⁻¹)</th>
<th>Brown Trout % YOY (percent)</th>
<th>Brown Trout YOY Biomass (g 0.1ha⁻¹)</th>
<th>Brown Trout Density (fish 0.1ha⁻¹)</th>
<th>Brown Trout % YOY (percent)</th>
<th>Brown Trout YOY Biomass (g 0.1ha⁻¹)</th>
<th>Rainbow Trout Density (fish 0.1ha⁻¹)</th>
<th>Rainbow Trout % YOY (percent)</th>
<th>Rainbow Trout YOY Biomass (g 0.1ha⁻¹)</th>
<th>Rainbow Trout Density (fish 0.1ha⁻¹)</th>
<th>Rainbow Trout % YOY (percent)</th>
<th>Rainbow Trout YOY Biomass (g 0.1ha⁻¹)</th>
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