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# Within-Basin Variation in the Short-Term Effects of a Major Flood on Stream Fishes and Invertebrates

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## ABSTRACT

We measured changes in invertebrate and fish abundance before and after a major flood event in the northeastern US, in three stream reaches that differed with respect to flood intensity. Fish and invertebrate abundance was lower at all three sites following the flood. The smallest change in fish and invertebrate abundance occurred in the site experiencing the lowest-magnitude flood (~ bankfull). The two remaining sites experienced overbank flooding and major changes in species abundances. Changes in abundance were greatest at the site experiencing greater geomorphic change (bedload movement and sedimentation), even though hydrologic intensity (velocity, shear stress, unit stream power) was greater at the other site. Aquatic invertebrate and underyearling fish abundances were substantially reduced at these sites, while overyearling salmonids exhibited normal or greater-than-normal abundance. Among invertebrates, abundances of baetid mayflies, which are multivoltine and disperse rapidly via drift, recovered more rapidly than other mayfly families. Our results reinforce the contention that geologic setting can strongly influence the short-term impact of floods. In addition, we provide direct evidence that particular species and age classes are resistant to even the most extreme floods expected in a given region.

## INTRODUCTION

Understanding the effects of large, infrequent floods on stream habitats and ecosystems is a major challenge for research. Opportunistic studies measuring species abundances before and after major flood events have yielded a wide range of results. In some cases, stream communities appear to be both highly resistant and resilient to major floods (Pusey et al. 1993, Lobon Cervia 1996, Dolloff et al. 1994). In contrast, the impacts of some extreme events appear to persist for years or decades (Murphy and Meehan 1991, Grossman et al. 1998). Two mechanisms are likely to

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produce such long-term effects. First, floods associated with major sediment inputs influence habitats by reorganizing channel structure in a manner that may persist until the next major event (Anderson and Calver 1977; Brunsden and Thornes 1979). Second, extreme events can affect populations by greatly increasing, or reducing, the strength of cohorts in long-lived species, such as many fishes (Jennings and Philipp 1994; Grossman et al. 1998).

Short-term changes in species abundances (e.g. a recruitment failure) may determine whether a flood event has important and persistent ecological consequences. While there is a general consensus that more intense floods generate greater immediate impacts (Resh et al. 1988), we know little about the specific thresholds of flood magnitude necessary to produce measurable effects on species abundances. This information is useful due to the highly predictable inverse relationship between flood magnitude and flood frequency; if thresholds of flood impact are known, it is possible to determine the frequency with which this level of impact will affect particular stream communities (Townsend and Hildrew 1994).

Between-reach variation in the intensity of a specific flood event may provide an opportunity to examine the relationship between flooding and short-term ecological response. Individual flood events are rarely uniform in intensity across drainage basins or geographical regions. As a result, the correspondence between flood intensity and ecological impact can be tested and used to define thresholds that result in discernable ecological effects. This type of study may be more powerful than studies that focus largely on differences before and after floods (Angradi 1998). Studies that only examine before- and after abundances have important limitations. High temporal variation in species abundances makes it difficult to unambiguously attribute differences observed in before and after studies to floods. In addition, change in abundance may be strongly influenced by flood timing, as well as by flood magnitude (Angradi 1998). Therefore, analyses that compare post and pre-event conditions across sites that differ with respect to flood intensity can control for seasonal differences and provide a more powerful method for assessing thresholds of flood impact (Stewart-Oaten et al. 1986).

This approach requires that a wide range of flood attributes be measured when ranking sites with respect to flood intensity. Relative differences among sites in flood magnitude may depend upon which flood attributes (i.e., discharge rise, shear stress, bed movement) are measured, and sites with higher impacts according to one set of criteria may be judged to have lower impacts with respect to another set of criteria. This approach also requires a whole-community-level analysis, as floods differentially affect different species and age classes. For example, larval and underyearling fish have been repeatedly shown to be more vulnerable to flooding displacement then older, larger, individuals (Pearsons et al. 1992, Chapman and Kramer 1991, Jensen and Jonsson 1999). Similarly, stream benthic invertebrates differ greatly in vulnerability to displacement by flooding, depending on their morphological, behavioral, and life-history characteristics (Lancaster and Hildrew 1993).

On July 2, 1998, heavy rainfall produced record flooding throughout northern and central regions of Vermont and New Hampshire, with many basins experiencing the flood of record. This area includes most of the White River basin, a major tributary of the Connecticut River draining 1790

$km^2$ of the east flank of the Green Mountains. The White River basin is the subject of several long-term studies of stream fish and invertebrate populations, and so we took advantage of this occurrence to assess whether effects on species abundances corresponded to quantitative estimates of differences in flood intensity among sites. Our specific objectives were to determine 1) if sites experiencing highest flood intensity experience the most change in species abundance and community structure following the flood, 2) which geomorphic and hydrologic attributes contributed to ecological effects, and 3) which species and age-classes were most influenced by flood impact.

## METHODS

### Study Sites and Ecological Surveys

We conducted the study in three stream reaches, each in a separate third order tributary in the White River basin. These tributaries (Tweed River, Hancock Branch, and Bethel-Gilead Brook) are typical upland cobble-bottomed, cold-cool water streams draining deciduous forests (Nislow et al. 1999). Fish communities consisted of three species of salmonids (brook trout *Salvelinus fontinalis*, rainbow trout *Onchorhynchus mykiss*, juvenile Atlantic salmon *Salmo salar*), two species of cyprinids (blacknose dace *Rhinichthys atratulatus* and longnose dace *Rhinichthys cataractae* ) and one species of sculpin (slimy sculpin *Cottus cognatus*). Invertebrate communities were typical assemblages dominated by the aquatic insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies).

Each tributary was represented by a single, 100 m study section, where fish, invertebrates, and physical habitat had been sampled in the previous year. Fish, invertebrate and habitat data were collected at these sites from two weeks to one year before the flood event, and from two weeks to nine months after the event. Aquatic macroinvertebrates were sampled at each study site by collecting nine individual Surber samples (350 μm mesh size) at 10 m intervals at randomly-selected river quarter points (0.25 of the total distance from the right or left bank), which were combined into a single, composite sample for each sampling bout. Each site was sampled twice pre flood and three times post flood. Samples were preserved in 70% ethanol. Invertebrates were sorted from a subsample (minimum necessary to achieve 100 individual invertebrates) of the original composite sample, identified to the level of family, enumerated, and measured using an ocular micrometer. Fish were sampled with three-pass removals of the entire 100 m study section, using a backpack electroshocking unit set to 500V, straight DC current (McMenemy 1995). We recorded the standard length and weight of all individuals, which were then returned to the streams. Population estimates were calculated using a modified Zippen maximum likelihood method (McMenemy 1995). At the time of sampling, young-of-the-year sculpins and dace were too small to effectively capture by electrofishing and were not included in the population estimates.

### Physical Habitat and Hydrologic Model

Flood intensity was assessed at the three sites using two techniques.

First, to assess geomorphic change, we measured active channel width and substrate size (using a modified Wentworth scale (Heggenes and Saltveit 1990), once pre-flood and once post-flood (the following year) at each site. These measurements were taken along bank-to-bank transects in a stratified random design, where transects were assigned randomly within 10 m strata from the downstream to upstream end of the study reach (Nislow et al. 1999). At each transect and for each sampling bout, channel width was measured and substrate was charaterized at 1m intervals. Second, to assess hydrologic intensity, we used HEC-RAS, a hydrologic model designed by the US Army Corps of Engineers (Hoggan 1989) to estimate key hydrologic and hydraulic parameters occurring during the flood. HEC-RAS calculates discharge and other hydraulic variables from field-derived channel data using a standard-step iterative process to reconstruct water surface profiles (cf. Hoggan 1989) and is based on the Manning Equation:

$$Q = AV = (A * R^{.67} * S_e^{.5})/n$$

where A = channel cross-sectional area, V = velocity, R = hydraulic radius (equal to depth in wide shallow channels), $S_e$ = energy slope, and n = Manning's roughness coefficient. In order to use HEC-RAS we needed to establish 1) the water surface elevation at the peak of the flood, and 2) the channel bed topography. To establish the flood water surface level we visited the sites immediately after the flood and searched for high water marks. We surveyed in three channel cross sections at the upstream, middle and downstream ends of each site, using an automatic level and a stadia rod, and established elevations of both high water marks and cross-sections relative to an arbitrary benchmark. We then ran the model to determine 1) the discharge necessary to achieve the measured flood elevations, and 2) the velocity, shear stress, and unit stream power associated with this modeled discharge.

Analyses

Our overall goal was to rank the sites with respect to flood magnitude, then test whether these ranks corresponded to ranks of short-term changes in fish and invertebrate abundance. As this was an opportunistic study, lack of true, consistent replication in time and space prevented our use of inferential statistics and limited analyses to qualitative comparisons. We used the hydrologic model results, combined with pre- and post-channel substrate and channel dimension data to determine which of the sites were subject to the most intense flood (see previous section). To assess flood impacts on abundance, we calculated the difference between pre- and post-abundances of individual taxa, and differences in community structure (see next paragraph). For individual taxa (and, in the case of salmonid fishes, separate age-classes), we express flood effects as a proportional difference, where

$$\text{Proportional Difference} = \frac{\text{Abundance(pre-flood)} - \text{Abundance(post-flood)}}{\text{Abundance(pre-flood)}}$$

By standardizing pre- and post-flood differences in abundance by pre-flood abundances, the use of proportional differences allows flood effects to be comparable across fish and invertebrate taxa that differ widely in initial, pre-flood abundance. Fish values are the differences between 1998 and 1997 values; invertebrate values are the differences between the mean values of the two pre-flood (May and June) vs. three post-flood (July, September, December) samples in 1998. In addition to taxa-specific proportional differences, we calculated total community dissimilarity between pre- and post-flood samples (= Euclidean distance), for both invertebrate and fish communities.

In all, we calculated thirteen separate measures of pre- and post-flood differences (Table 1). These measures were then used to test two separate hypotheses regarding flood impact.

H1: Sites with the highest flood intensity (based on hydrologic and geomorphic data) will exhibit the largest change in abundance and community similarity pre- and post-flood.

H2: Taxa that are most vulnerable to floods (based on body size and dispersal characteristics) will exhibit largest negative changes in abundance post-flood.

Table 1. Ranks of pre- vs. post-flood differences in thirteen abundance parameters in the three study sites, along with mean and median ranks for all abundance parameters. Ranks range from 1 (smallest change) to 3 (greatest change).

| Parameters | Sites | | |
|---|---|---|---|
| | Tweed River | Hancock Branch | Bethel-Gilead Brook |
| Overyearling Atlantic salmon | 3 | 2 | 1 |
| Age-0 Atlantic salmon | 1 | 2 | 3 |
| Overyearling brook trout | . | 2 | 3 |
| Age-0 rainbow trout | 1 | 3 | 2 |
| Overyearling rainbow trout | . | 2 | 3 |
| Total age-0 salmonids | 1 | 2 | 3 |
| Total overyearling salmonids | 2 | 1 | 3 |
| Longnose dace | 1 | 2 | 3 |
| Blacknose dace | 1 | 2 | 3 |
| Sculpin | 3 | 1 | 2 |
| Fish community similarity | 1 | 2 | 3 |
| Invertebrate abundance | 1 | 2 | 3 |
| Invertebrate community change | 1 | 2 | 3 |
| Mean Rank | 1.45 | 1.92 | 2.69 |
| Median Rank | 1 | 2 | 3 |

To test Hypothesis 1, we ranked each proportional difference, from 1 (smallest difference) to 3 (largest difference), and then compared differences in the abundance changes. To test Hypothesis 2, we classified fish and invertebrates into groups predicted to be more or less vulnerable to flood impact, and then compared pre- and post-flood abundance changes between groups. For fish, based on the results of previous studies (Pearsons et al. 1992, Chapman and Kramer 1991, Jensen and Jonsson

1999) we predicted that the abundance of overyearling salmonids (> 80mm long) would show a smaller reduction in abundance following the flood than would underyearling salmonids and non-game species (< 80mm long). For invertebrates, we focused on differences in response between families of mayflies (Order Ephemeroptera). Mayflies were the most abundant and diverse insect order in our invertebrate samples and therefore provided the best opportunity to test for this effect on invertebrates. We predicted that mayflies in the family Baetidae would show a smaller reduction in abundance and a more rapid recovery than would families Ephemerellidae, Heptageniidae, and Leptophlebiidae. We made this prediction for two reasons. First, many baetids are multivoltine in this region, with a summer generation that could more rapidly restore population levels than could the other, univoltine, mayfly families. Second, baetids show active entry into the drift (Rader 1997) and are very effective dispersers in part due to this facility, which could also facilitate recovery from flood events.

## RESULTS

### Flood Intensity at the Three Sites

It was possible to rank the three sites with respect to the relative intensity of the flood event. By both geomorphic and hydrologic criteria, the Tweed River site experienced markedly lower flood intensity than the other two study sites. This site experienced an approximately bankfull discharge, corresponding to a two-year recurrence interval, based on both visual inspection of the channel, and on regional curves relating bankfull discharge to drainage area (Magilligan, unpublished data).

In contrast, the Hancock Branch and Bethel-Gilead Brook sites experienced significantly greater than bankfull discharge conditions and correspondingly significantly greater velocity, shear stress, and stream power than did the Tweed River site (Fig. 1). While Hancock Branch and Bethel-Gilead Brook experienced similar discharges, they differed significantly with respect to other criteria. The Hancock Branch site had markedly higher bed shear stress, stream power and channel velocity (Figure 1) than the Bethel-Gilead site. However, we did not observe any

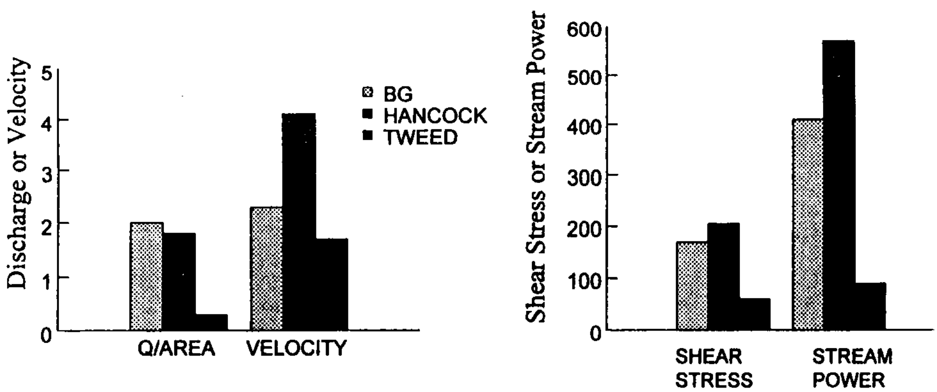


Figure 1. HEC-RAS estimates of hydrologic and hydraulic parameters in three study tributaries during the flood of 1998. Estimates are based on river stage measured by post-flood high water marks.

evidence of major bed movement or bar formation at the Hancock Branch site, and both channel widths and Wentworth substratum scores were essentially unchanged before and after the flood (Fig. 2). In direct contrast, although the Bethel-Gilead Brook site experienced lower shear stress and velocity than the Hancock Branch, there were major bed movement and channel changes after the flood. Large-scale bed movement deposited a large cobble bar over most of the pre-existing channel, significantly narrowing the active channel. In addition, this site experienced a major influx of fine sediments, significantly lowering Wentworth scale substratum scores compared to pre-flood conditions.

These combined results indicated two alternative orderings of sites with respect to flood intensity. On the basis of hydrologic intensity, the Hancock Branch site was predicted to show the largest differences between pre- and post-flood abundances. Alternatively, on the basis of geomorphic change, the Bethel-Gilead site was predicted to show the largest difference pre- and post-flood.

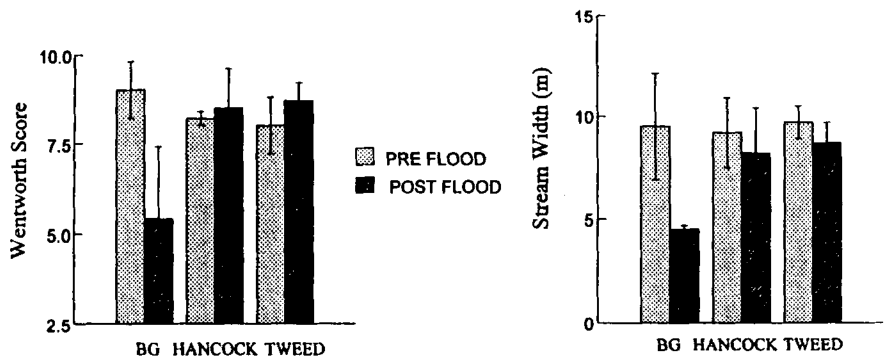


Figure 2. Mean (S.E.) channel width and substratum conditions (Wentworth substrate size score) in three study tributaries before and after the flood of 1998; n = 5 transects per study plot.

Invertebrates

Invertebrate abundances were generally lower following the flood, but changes differed among sites (Figure 3a). There was essentially no reduction in abundance at the Tweed River site, where differences were well within the range of seasonal variation in abundance. Reduction in overall abundance, averaged over all pre- compared to all post-flood samples, was greatest at the Bethel-Gilead Brook site, where it was almost an order of magnitude lower post-flood. Hancock Branch changes were intermediate between those of the Bethel-Gilead Brook and Tweed River sites. Similarly, overall community change (as measured by Euclidean distance between pre- and post-flood samples) was greatest at the Bethel-Gilead Brook site, and very similar between the Hancock Branch and Tweed River sites (Figure 3b).

In addition to differences among sites, we observed differences with respect to changes in abundance for different invertebrate taxa. Specifically, mayfly families predicted to be more vulnerable to floods showed larger decreases in abundance then did mayflies in the family Baetidae, which were predicted to be more resistant to floods (Figure 4). Other families (Ephemerellidae, Heptageniidae, and Leptophlebiidae) declined in all three sites. All three of these families were absent from samples collected one week after the flood, and abundance only recovered at the Tweed River site. In contrast, baetid abundance was only reduced at the Bethel-Gilead Brook site, recovered rapidly, and by December, was essentially back to pre-flood levels in all sites.

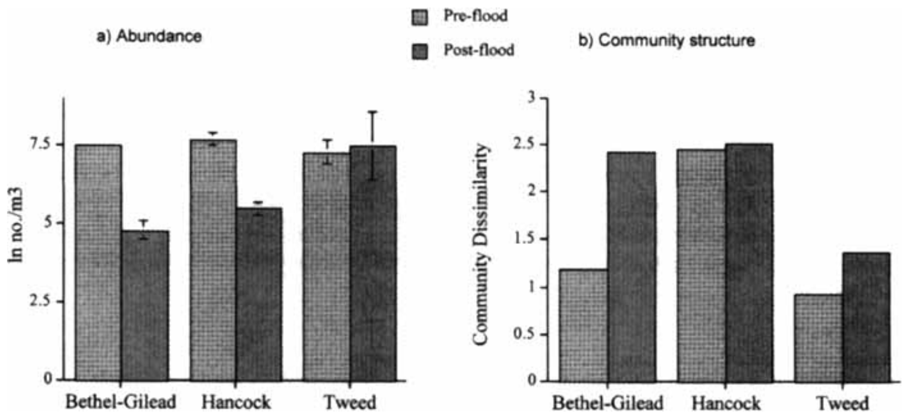


Figure 3. a) Benthic invertebrate abundance (ln no./0.33 $m^2$) in the three study tributaries before and after the flood of 1998, and b) Euclidean distance between pre- vs. post-flood invertebrates assemblages.

Fish

With respect to fish communities, abundances of age-0 salmonids and non-game species were generally reduced following the flood. The greatest reduction was seen at the Bethel-Gilead Brook site and the smallest reduction at the Tweed River site (Figure 5). This change was most striking for age-0 Atlantic salmon at Bethel-Gilead Brook. In previous years, this stream had one of the highest abundances of age-0 salmon in the entire White River basin (Nislow et al. 1999). In 1998, after the flood, no age-0 salmon were found in the study reach. Age-0 rainbow trout also had much lower abundances in all three sites following the flood, while age-0 brook trout, which were not very abundant in any of the three sites, declined considerably at Bethel-Gilead Brook but showed little difference in the other two sites. In strong contrast to the smaller fishes, overyearling salmonids increased at all three study sites (Figure 5). Overyearling salmonid densities were highest at the Bethel-Gilead Brook site, where we observed very high densities of both overyearling salmon and rainbow trout.

Relationship Between Flood Intensity and Abundance Changes

The magnitude of changes in species abundances and community similarity was related to flood intensity, as measured by geomorphic change, but not hydrologic intensity. For the 13 taxa and community measures, the Bethel-Gilead site had a higher rank-score then either the Hancock Branch or the Tweed River site (Table 1). This indicates that for each comparison, the Bethel-Gilead Brook site tended to have the greatest differences (10 of 13 comparisons) between pre- and post-flood values, strongly supporting the geomorphic change ranking. In contrast, the Hancock Branch had the greatest pre-post difference in only one of 13 comparisons. Therefore, the hydrologic intensity ranking was strongly rejected.

## DISCUSSION

Establishing the relationship between flood intensity and ecological response is key to understanding the structure and function of stream ecosystems. We found that ecological impact, as measured by changes in abundance, consistently tracked flood intensity among sites. Ecological impact was least at the site where flood levels only reached the bankfull flood stage, greatest at the site where overbank flows and major channel change occurred, and intermediate at the site experiencing overbank flows but where no channel change occurred. Our combination of hydrologic modeling and simple channel surveys provided a rapid, powerful method for partitioning hydrologic and geomorphic aspects of floods. This approach

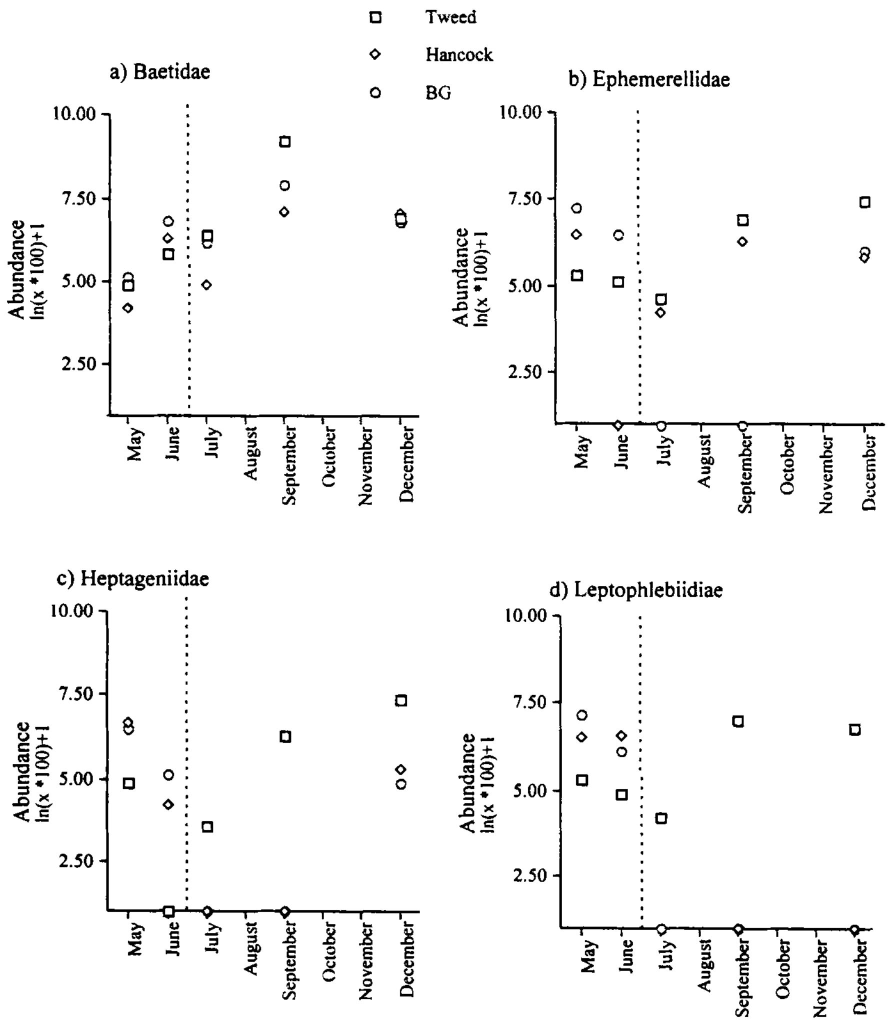


Figure 4. Abundance of four mayfly families (a-d) (ln no./m2) in the three sites over the course of the study year. Stippled vertical lines separate pre-flood from post-flood samples.

should aid efforts to extend the spatial extent of flood studies and help elucidate mechanisms of flood impact on stream communities.

Our results indicate that hydrologic and hydraulic measures of flood intensity were insufficient, on their own, to predict the magnitude of change in species abundances. The two sites experiencing similar flow volumes, Bethel-Gilead Brook and Hancock Branch, translated this event into significantly different impacts on habitats and biota. Greater impacts were observed where shear stresses were lower, but considerable bedload movement occurred. Previous studies are divided as to the relative importance of substrate movement vs. bed shear stress as a mechanism for

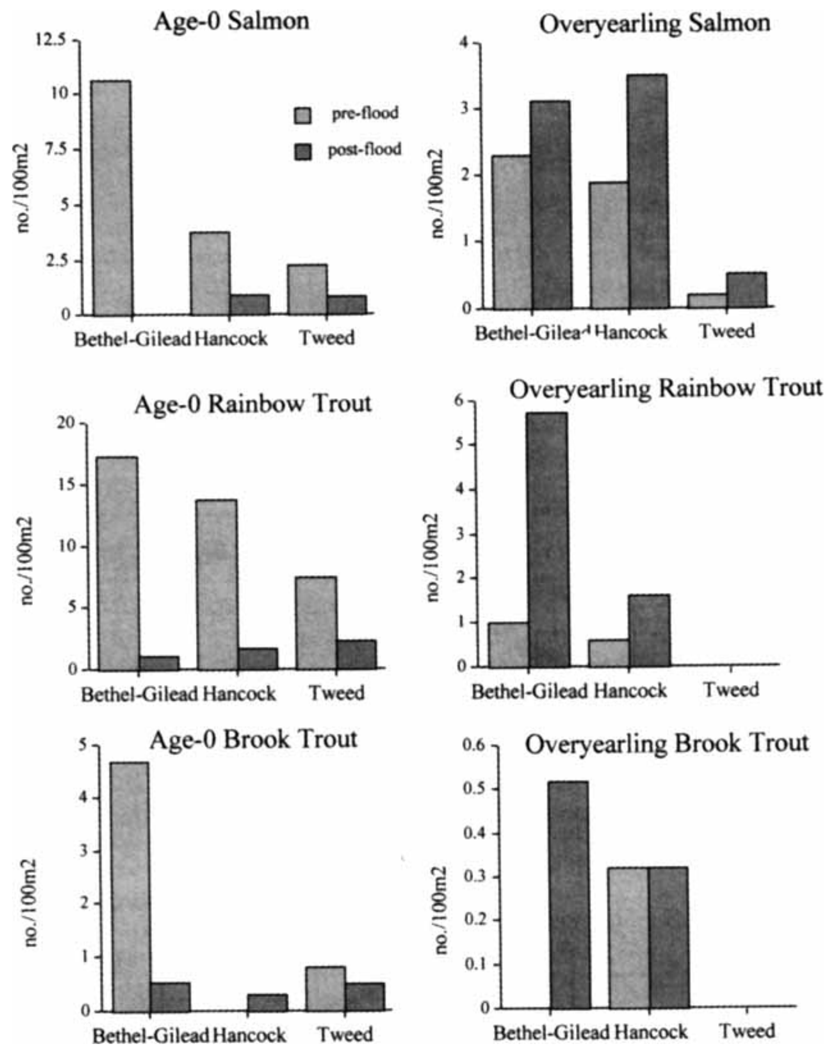


Figure 5. Abundance (no./m2) of overyearling and underyearling salmonids (combined brook trout, rainbow trout, and Atlantic salmon) in the three study tributaries before and after the 1998 flood. Bars projecting above the zero-line indicate greater abundance post-flood; bars projecting below the zero-line indicate reduced abundance post-flood.

the displacement of individual organisms (Resh et al. 1988, Lancaster and Hildrew 1993). Our results are consistent with those obtained by Cobb et al. (1992) and Biggs et al. (1999), and indicate that bed movement appears to be significantly more important than high shear stress in the loss of individuals following a flood.

Differences in geologic and geomorphic setting between the two sites may hold the key to differences in flood impacts. Specifically, glacial deposits associated with the Bethel-Gilead Brook site provided a source of material that enabled the flood to dramatically reconfigure the stream channel. In contrast, bedrock on the channel at the Hancock Branch site promoted channel stability and appeared to lessen effects of the floods on sediment and bedload transport, significantly reducing the impacts on biotic communities. In the New England region, where the thickness of overlying glacial till shows considerable variation both within and among streams (Denny 1982), geologic setting may strongly determine the immediate ecological impacts of floods.

In addition to differences among sites, flood effects varied among aquatic taxa. Consistent with a number of recent studies (Chapman and Kramer 1991, Pearsons et al. 1992, Hayes 1995), size, age, and life history traits appeared to have a major influence on vulnerability to flooding. For invertebrates, greater resistance and resilience of mayflies in the family Baetidae, compared to other mayfly families, was consistent with previous information on the dispersal and life-history traits of mayfly families. For fish, small underyearling salmonid and non-game fish densities were severely reduced, while we found no negative effects on large, overyearling salmonids. By directly measuring impacts of an event of known recurrence interval, our results also provide direct evidence that overyearling salmonids may be essentially unaffected by even the most extreme floods (Lobon-Cervia 1996, Letcher and Terrick 1998).

High overyearling salmonid densities at the heavily-impacted Bethel-Gilead Brook site suggest that major floods may have some positive effects on aquatic communities in these systems. Mechanisms by which major floods can positively impact stream fishes include negative effects on introduced predators that favor native fishes and reduction of interspecific competition resulting in increased growth and maturation (Letcher and Terrick 1998). In our case, increased overyearling salmonid abundance may have been the result of habitat change. For example, at the Bethel-Gilead Brook site, deposition of a large cobble bar narrowed the channel by ~ 50%. This resulted in a deeper, more stair-stepped profile with numerous scour pools, habitats preferred by overyearling salmon and trout (Bjornn and Reiser 1991). Habitat change at this site therefore appears to be more important to overyearling abundance than any negative impact of the flood.

This potential mechanism is particularly interesting, since landscape change (i.e., large-scale deforestation) (Foster 1992) in the New England region is thought to have generally diminished the frequency and quality of pool habitat in upland streams (Roy[a], pers. comm.) by removing the source of large woody debris (LWD) and progressive channel widening and flattening under unstable bank conditions. Although the region is now largely reforested (Foster 1992), large infrequent floods may be required to restore instream habitats to pre-disturbance conditions. Although we did

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not quantify LWD before or after the flood in these streams, we did observe that large amounts of LWD entered all of the streams after the flood, an observation also made for southern Appalachian (Dolloff et al. 1994) and Sierra Nevada (Berg et al. 1998) streams. The long intervals between major floods capable of reshaping channel dimensions and recruiting LWD may therefore influence the time course of habitat and biotic community recovery from major landscape change (Harding et al. 1998).

While the relationship between flood intensity and abundance change found in this study is interesting and suggestive, these conclusions must take into account some major limitations. As in many opportunistic studies of flood effects, lack of adequate pre-flood sampling, small number of study sites, and inadequate replication within streams must qualify the generalities of our findings. Therefore, we view the major importance of this work as providing a methodology and an approach that will aid future understanding of the role of flood disturbance in stream ecosystems.

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