The Effect of Insect Mortality and Other Disturbances on Water Yield in the North Platte River Basin

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The Effect of Insect Mortality and other Disturbances on Water Yield in the North Platte River Basin

C.A. Troendle and J.M. Nankervis

1. Introduction

The USDA Forest Service, Rocky Mountain Region, and the U. S. Bureau of Reclamation contracted for a series of analyses of water yield in the Platte River Basin beginning in 2000. The analyses were completed in two phases, first for the North Platte Basin (Troendle and Nankervis 2000, Troendle et al., 2003) and then for the South Platte Basin (Troendle et al., 2006). In total, these reports defined the potential effects of past Forest Service activities on the stream flow regime of the Platte River Basin. It was documented that forest management activities were not particularly extensive and hydrologic response to those activities appears to have been minimal. Fire effects, such as what occurred following the Hayman Fire and the threat of insect infestation and mortality from the mountain pine beetle and spruce beetle, appear to have a far greater potential effect on water yield. This report describes the results of a study designed to revisit these earlier efforts to evaluate the long-term impacts of the recent insect infestations on water yield from the North Platte River Basin. Insect driven tree mortality has proven to be severe and extensive in both the lodgepole pine, spruce, and other forest types throughout the west and it can be expected that the hydrologic consequences will also be significant and extensive. The primary question in this study is “what has been the effect of recent insect mortality in the forest of the North Platte River basin on water yield from the North Platte River above Seminoe Reservoir?”

In an attempt to answer that question, this report:

1) Reviews the documented effects of forest disturbance, whether from timber harvest, fire, or insect mortality on water yield.
2) Updates and provides the results of a study designed, in 2003, to document future effects of catastrophic disturbance, such as wide spread insect infestation, on water yield based on documented responses at 8 pre-selected USGS stream gauging sites in both the North and South Platte River Watersheds.
3) Characterizes the effects of insect mortality on vegetation in the 8 study watersheds as well as in the North Platte River above Seminoe Reservoir, relates those changes in vegetation cover to observed changes in water yield from the 8 study watersheds, and uses the estimates of vegetation change to model the hydrologic consequences in both the 8 study watersheds and for the North Platte River watershed contributing to the Seminoe Reservoir.
4) Projects the effects of insect mortality on stream flow contributions to the Seminoe Reservoir for 60-years into the future.
2. Effect of Forest Disturbance on Water Yield

Over time, the continuity equation defining water yield can be simplified to the expression:

\[
\text{Runoff (Water Yield)} = \text{Precipitation (P)} - \text{Evapotranspiration (ET)} \quad (\text{Eq. 1})
\]

It is well recognized that the variability of runoff, in time and space, is most strongly influenced by the variability in precipitation (P). Evapotranspiration (ET), the sum of all evaporative losses from interception, transpiration, and soil water evaporation is the less variable component in the balance and codependent on precipitation as well as other factors such as energy distribution and vegetative cover. Change in soil water storage (+/-) is often considered in the equation for short time periods such as seasonal or annual balances, but over time it is not a significant factor in the equation. Based on studies at the Fraser Experimental Forest, in the upper Colorado River Basin, Troendle and Reuss (1997) found that on average annual

\[
\text{Evapotranspiration (ET) = 460mm + 0.28 * (Precipitation – 460mm).} \quad (\text{Eq. 2})
\]

This relationship implies that approximately 460 mm (18.1 in.) of precipitation is needed to meet basic evaporative demands before significant amounts of runoff will be generated. Once the threshold is exceeded, ET increases by 0.28 mm for each mm of precipitation in excess of 460 mm. Although developed based on data from the upper Colorado River Basin, this relationship is consistent with other regional estimates (Strahler 1966; Troendle and King 1985). In application, site specific factors such as aspect, elevation, species composition and vegetation cover, as well as precipitation amount, form, and distribution modify this relationship.

Precipitation increases with elevation and most runoff occurs from higher elevation lands where annual precipitation exceeds 460 mm and is more than adequate to meet ET demands, support forest vegetation, and generate runoff. In turn, forest vegetation is a significant water user and changes in forest cover, whether increases or decreases due to timber harvest, fire, insect and disease mortality, re-growth or in-growth all have the potential to influence the magnitude of ET and therefore water yield. Most of our understanding about the general effect of vegetation disturbance on water yield comes from a myriad of paired watershed experiments worldwide, as well as from complimentary plot and process research designed to help understand the watershed processes affected.

2.1. Timber Harvest

There have been many studies worldwide that demonstrate changes in forest cover result in a change in water yield. Hibbert (1967), Troendle and Leaf (1980), Bosch and Hewlett (1982), Stednick (1996), and more recently Ice and Stednick (2004) have summarized the findings from some or all of these studies. MacDonald and Stednick (2003) provided a review more specific to Colorado. Bosch and Hewlett (1982) noted that although stream flow response to a change in forest cover is strongly related to climate and species composition there is a universal relationship that reductions in vegetation cover cause increases in stream flow (Figure 1).
Bosch and Hewlett (1982) arbitrarily divided the paired watersheds studies conducted worldwide into 3 response classes; conifer forests, deciduous forests, and shrub land. As a general observation, they found that the largest percentage change in annual water yield following reductions in vegetation cover occurred in conifer forests while the least relative response occurred following modification of scrub land cover (Figure 1). The differences in response between the vegetation types largely reflected differences in the water use characteristics of the plant species as well as differences in the precipitation and energy at the sites on which they occur. Although the data points are not plotted on Figure 1, the variability is quite high as indexed by Coefficient of Determination’s (R²) for the three regressions shown in Figure 1 that range from a high of 0.42 for conifers to a low of 0.12 for scrub land vegetation (Bosch and Hewlett 1982). Higher elevation, colder, wetter sites tend to support conifers while warmer, drier sites tend to support grasses and shrubs. It is also a universal observation that the magnitude of the water yield change that occurs in response to alteration of the vegetation increases with increasing precipitation (Bosch and Hewlett 1982; Troendle 1983). Wet years result in a greater change in water yield for a given alteration in vegetation than occurs in dry years. This implies that for a given watershed, the response in water yield will be greater in wet years than in dry years. It also implies that for a given treatment, watersheds receiving a greater annual precipitation will generally produce the larger response.

Scaling back to the data from the 95 paired watershed experiments conducted in the United States demonstrates that, on average, stream flow increases by nearly 2.5 mm (0.1 in.) for each 1 percent of the watershed area harvested (Stednick 1996). Because annual water yield varies greatly from year to year, due to variability in annual precipitation, the general conclusion is that approximately 20 percent of the vegetation, above the point of stream flow measurement, must be removed before a statistically significant change in flow is likely to be detected (Hibbert, 1967; Bosch and Hewlett 1982; and Stednick 1996). As Bosch and Hewlett (1982) suggest, however, reducing forest cover by less than 20 percent may well produce statistically non-significant increases in stream flow that presumably approach zero increase at zero change in forest cover.

On a regional scale, the Rocky Mountain region is fortunate in having a series of carefully controlled paired watershed experiments intended to evaluate the effects of forest harvest on water yield. The first such experiment was conducted at Wagon Wheel Gap in south central Colorado on the headwaters of the Rio Grande River (Bates and Henry 1928; Van Haveren 1981; Troendle and King 1987).

This study was followed by a series of watershed studies in the region (Hoover and Leaf 1967; Troendle and Leaf 1981; Troendle and King 1985; Troendle and King 1987; Troendle et al., 2001; Stednick and Troendle 2004), with the longest running and most detailed study having been conducted on the Fool Creek watershed on the Fraser Experimental Forest in central Colorado (Hoover and Leaf 1967; Troendle and King 1985). At Fool Creek, mean annual precipitation is estimated to average about 760 mm (29.9 in.) per year while estimates of annual ET range from 450 to 570 mm (17.7 to 22.4 in.) per year and are dependent on precipitation amount and energy (Troendle and King 1985), indicating that 60-75 percent of the annual precipitation input is lost through the components of ET.
Figure 1. Reduction in vegetation cover and increase in streamflow (redrawn from Bosch and Hewlett 1982).

Approximately 40 percent of the Fool Creek watershed, or 50 percent of the commercially forested area, was clearcut during 1954 to 1956 in a pattern of alternating cut and leave strips. Comparison of the “average” 15-year pre- and 15-year post-treatment hydrographs for Fool Creek (Figure 2) clearly show that, on average, forest harvest increased both total seasonal flow and peak flow. The relative difference between the pre- and post-treatment hydrographs for Fool Creek is typical of the response observed on all paired watershed experiments addressing the effect that forest disturbance has on water yield in the cold snow zone of the central and northern Rocky Mountains (Bates and Henry 1928; Swanson and Hillman 1977; Swanson et al., 1986, Troendle and King 1987; Troendle and Bevenger 1994; Troendle and Reuss 1997; Troendle et al., 2001).

The “average” first year response to timber harvesting on Fool Creek was equivalent to a 100.0 mm (3.9 in.) increase in seasonal water yield from the entire watershed area along with a 22 percent increase in average peak flow (Troendle and King 1985). During the 1956 to 1983 post treatment period, observed increases in stream flow ranged from a high of 162 mm (6.4 in.) in the wettest year to a low of 36 mm (1.4 in.) in the driest year. On average, stream flow increases reflect a 50 percent reduction in the annual ET that would have occurred on the clearcut portion of the watershed had it not been harvested (Troendle and King 1985; Troendle and Reuss 1997). By 1983, 28 years after harvest, a significant decline in stream flow resulting from forest regrowth could be detected. A more recent analysis indicates that complete hydrologic recovery occurred in about 60 years (personal communication Kelly Elder, Rocky Mountain Research Station). As can be inferred from Figure 2, the most notable increase in flow occurs in May. Only occasionally was a significant increase detected during the month of June (Troendle and King 1985). No detectable effect has been documented on recession flows occurring from July to October (Troendle and King 1985).
There is little opportunity for measurable increases in water yield to occur during the growing season because summer ET is water limited. Less than 5 percent of summer precipitation is returned as storm flow, indicating that the water is retained and used on site (Garstka et al., 1958; Troendle and King 1985; Bevenger and Troendle 1987), and that a reduction in summer ET does not appear to detectably influence stream flow response during the current year.

Figure 2. Average hydrographs from the Fool Creek Watershed for the 15 year period before and after timber harvest (Troendle and King 1985).

Changes in ET, following forest removal in the cold snow zone, are the result of both a net reduction in growing season soil moisture depletion by residual forest vegetation and a decrease in winter and summer interception and evaporative losses (Wilm and Dunford 1948; Dietrich and Meiman 1974; Potts 1984b; Troendle and Meiman 1986; Troendle 1987; Troendle and Reuss 1997). Although there is a net reduction in evaporative losses resulting in a net increase in stream flow, some evaporative components increase following harvest. In the cold snow zone, precipitation accumulates over the winter as snow pack, with minimal ablation over the accumulation period. When the snowpack begins to melt in spring, the melt water first recharges the soil by replacing the soil water that was used during the previous growing season. Once soil moisture storage requirements are satisfied, subsequent melt water is available to become stream flow. At Fool Creek, which is comprised mostly of east and west facing slopes, approximately 30 percent of the increase in seasonal water yield that was observed following timber harvest (Figure 2) can be attributed to the decrease in winter interception losses from the canopy and resultant increase in the amount of water contained in the snowpack at the time of melt. Approximately 50 percent of the increase in water yield during the current water year can be attributed to the reduction in ET losses and soil water depletion during the previous summer. Reduced transpiration from the previous year leaves the soil wetter going into the winter period and reduces the soil moisture recharge requirement during the subsequent spring melt period.
Because less melt water is required to recharge the soil, runoff begins earlier following harvest with most of the flow change occurring early in the runoff period as depicted in Figure 2. The remaining 20 percent of the observed increase in water yield on Fool Creek can be attributed to the reduction in ET losses during the April and May period (Troendle and King 1985). Further reductions in ET occur during the balance of the growing season but these reductions are usually reflected in increased soil water reserves rather than demonstrated changes in water yield. In a nearby plot study investigating the mechanisms of stream flow generation, Troendle (1987) documented that the lateral movement of subsurface flow was initiated as much as a month earlier from a clear-cut hill slope when compared to the response from the adjacent uncut control plot. In addition, higher than normal precipitation in the fall also resulted in measureable lateral flow from the harvested hill slope while no lateral flow was initiated on the control hill slope, again a testament to the presence of wetter soils on the harvested slope.

Reduced winter interception losses can represent more than 50 percent of the increase in flow following harvesting on north slopes (Troendle and Meiman 1986; Troendle and King 1987) while representing only 20 percent of the potential flow change on south slopes. On north facing slopes, snow often remains in the canopy continuously from November to May (Hoover and Leaf 1967) and although the evaporation rate is low, the tremendous surface area of the intercepted snow allows a great deal of evaporative loss to occur prior to timber harvest. In contrast, on south facing slopes, intercepted snow more quickly falls from the less dense and warmer forest canopy thus allowing less opportunity for evaporative loss. The 130- to 190-mm (5.7 to 7.5 in.) reduction in summer ET that occurs on all aspects following clear cutting, as indexed by reduced soil water depletion, appears to be independent of aspect and dependent only on precipitation amount and distribution (Troendle 1987). The components of the ET losses partitioned through plot and process studies document the fact that the subalpine ecosystem is energy limited in winter and water limited in summer with aspect and vegetation density being a less significant factor in evaporative loss during the summer than in the winter. Opportunities for water yield change following forest disturbance are greater on north slopes than either east/west or south slopes.

As noted, there have been a number of paired watershed experiments conducted in the central Rocky Mountains that have demonstrated increases in flow following timber harvest. The study sites include Wagon Wheel Gap in south central Colorado (Bates and Henry 1928), Dead Horse Creek in central Colorado (Troendle and King 1987), Coon Creek in southern Wyoming (Troendle et al., 2001), and the Sturgis watershed in South Dakota (Anderson 1980; Troendle 1987b). There are also a few anecdotal observations on the effects of tree mortality due to insect mortality in northwestern Colorado (Love 1955; Bethlahmy 1974), wildfire in northern Wyoming (Troendle and Bevenger 1996), and beetle mortality in southwestern Montana (Potts, 1984a). Summary data for the experimental watershed studies are presented in Table 1. It should be noted that the annual flow increase listed in Table 1 represents the average increase in flow that occurred either during the first 5 years following treatment or for the entire post-treatment, whichever was longer.
The increases in stream flow following timber harvest presented in Table 1 are strongly correlated with the reduction in forest cover (Figure 3). Much of the variability in response that was observed by Bosch and Hewlett (1982) is eliminated when the responses are regionalized (Troendle and Leaf 1980). The Coefficient of Determination ($R^2$) associated with the regression of Annual Flow Increase over Percent Reduction in Cover (Figure 3) for the watershed data presented in Table 1 is 0.77. This is in contrast to the 0.42 obtained by Bosch and Hewlett (1982) for conifer sites worldwide.

All of the controlled watershed experiments included in either Table 1 or plotted in Figure 3, have involved partial clear cutting or thinning that impacted between 10 and 40 percent of the forest cover at the watershed level. Study results indicate that on average during the first 5 years following timber harvest, water yield was increased by nearly 70 mm (2.4 in.) as the reduction in forest cover increased from 10 to 40 percent (Figure 3). This represents a 2.33 mm (0.09 in) increase in water yield for each 1 percent reduction in forest cover within that range. This is consistent with the average of 2.5 mm (0.1 in.) increase per 1 percent of the watershed harvested that was calculated for all 95 paired watershed experiments in the United States (Stednick 1996). When possible, the 5 year average increase in water yield was used in developing the relationship in Figure 3 to help average the impact of precipitation variability. For example, 1957 was the first year following harvest on Fool Creek and it has proven to be one, if not the, wettest year on record. Nineteen seventy-seven, the first year following treatment on the North Fork of Deadhorse Creek was one of the driest years on record. Using the 5 year average response tends to mitigate the effect of precipitation variability on the change in water yield but not long enough to allow the change in water yield to be influenced by vegetation recovery.

Much of the variability in the average annual flow and the annual flow increase presented in Table 1 is accounted for by the differences that exist in the amount of precipitation that falls on the watersheds. As a means of demonstrating the relative response to timber harvest that occurs as a function of mean annual precipitation, the 5 year average annual flow increase in water yield for each watershed experiment (from Table 1) was divided by the percent reduction in cover that occurred on that watershed (from Table 1). The resultant was multiplied by 100 to arrive at an estimate of the relative increase in water yield that would have occurred had there been a 100 percent reduction in cover, or a complete clearcut on each watershed. For example, Fool Creek had an average annual flow increase of 115 mm as the result of a 40 percent reduction in cover. The increase in flow, expressed in terms of clearcut area would be equivalent to a 287.5 mm (115 mm increase / 40 percent * 100) increase in annual flow had timber harvesting reduced the percent cover to 0.0 on the entire watershed.

The purpose for normalizing, or transforming, the observed changes in annual flow to estimate what might have been the equivalent response had the entire watershed been clearcut is to help isolate the role precipitation plays in the observed flow change. The average annual flow increases are significantly influenced by the variation in precipitation between watersheds (Figure 4). Figure 4 demonstrates that the magnitude of the change in stream flow that results from a common timber harvesting practice (in this case a clearcut) varies from watershed to watershed in response to precipitation differences between the watersheds.
There is significant variability in the amount and distribution of precipitation that falls on the forested portions of the central and northern Rocky Mountains and the role that precipitation variability has on stream flow response following forest disturbance has to be considered. This is critical in extrapolating the changes in stream flow observed on the experimental watersheds to other sites and other conditions. It is evident from the relationships presented in Figures 3 and 4 that precipitation differences between watersheds, and from year to year, may have an equal or greater effect on the stream flow response to timber harvest, or forest disturbance, than the percent reduction in cover. This variability becomes more important when comparing plot responses in one location with those from another location within the watershed.

### Table 1.
Summary data for the documented paired watershed studies conducted in conifer forests in Colorado, Wyoming, and South Dakota

<table>
<thead>
<tr>
<th>Controlled Experiment Watershed</th>
<th>Location</th>
<th>Average annual Precipitation mm (in.)</th>
<th>Average Annual Flow mm (in.)</th>
<th>Reduction in Cover (%)</th>
<th>Annual Flow Increase mm (in.)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fool Creek</td>
<td>Colorado</td>
<td>762 (30.0)</td>
<td>367 (14.4)</td>
<td>40</td>
<td>115 (4.5)</td>
<td>Troendle and King 1985</td>
</tr>
<tr>
<td>Deadhorse Main</td>
<td>Colorado</td>
<td>526 (20.7)</td>
<td>286 (11.3)</td>
<td>10</td>
<td>18 (0.7)</td>
<td>Troendle and King 1987</td>
</tr>
<tr>
<td>Deadhorse North</td>
<td>Colorado</td>
<td>544 (21.4)</td>
<td>218 (8.6)</td>
<td>36</td>
<td>65 (2.6)</td>
<td>Troendle and King 1987</td>
</tr>
<tr>
<td>Deadhorse North Slope</td>
<td>Colorado</td>
<td>569 (22.4)</td>
<td>258 (10.2)</td>
<td>40</td>
<td>91 (3.6)</td>
<td>Troendle and King 1987</td>
</tr>
<tr>
<td>Coon Creek</td>
<td>Wyoming</td>
<td>870 (34.3)</td>
<td>542 (21.3)</td>
<td>24</td>
<td>76 (3.0)</td>
<td>Troendle et al., 2001</td>
</tr>
<tr>
<td>Sturgis Watershed</td>
<td>South Dakota</td>
<td>745 (29.3)</td>
<td>226 (8.9)</td>
<td>25</td>
<td>58 (2.3)</td>
<td>Anderson 1980</td>
</tr>
</tbody>
</table>
**Figure 3.** Reduction in forest cover and the first 5-year average increase in streamflow following harvest on paired watersheds in the central Rocky Mountains (Troendle and Porth, in draft).

**Figure 4.** Five year annual average water yield increase normalized to represent equivalent clearcut response (Troendle and Porth, in draft).
2.2. Other Disturbances

The paired watershed studies identified in Table 1 and depicted in Figures 3 can be complimented by two additional studies that indicate reductions in forest cover due to insect infestation (Love 1955) and fire (Troendle and Bevenger 1996) also result in proportional increases in water yield (Figure 5). In addition, the increase in water yield estimated to have occurred following complete clear cutting of one of a calibrated pair of forested hill slope run-off plots on the Fraser Experimental Forest (Troendle and Reuss 1997) is also consistent with the observations on the experimental watersheds. The increases in water yield for the 3 additional sites compares favorably and are not significantly different than the responses observed for the experimental watersheds (see Figure 5). In a re-analysis of the White River beetle kill data, Bethlahmy (1975) also noted that the largest increases occurred on the north facing portions of the watershed. Although limited in extent, the documented water yield increases following forest disturbance appears to be consistent, proportional, and dependent on the degree of disturbance, regardless of the cause of disturbance (Figure 5). The regression line and confidence bands presented in Figure 5 are based on the paired watershed data presented in Figure 3. All other information is presented for comparative purposes and was not used in the fitting process in any way. It would appear that insect mortality and fire (assuming no alteration in flow path) can have an effect on stream flow that is similar to timber harvest. It would also appear that the relationship between increases in water yield and reductions in forest cover, as presented in Figure 3, appears to be linear throughout the range of observations. Both Bosch and Hewlett (1982) and Stednick (1996) concluded that increase in flow following forest disturbance was linearly correlated with forest cover percent reduction as the relationship presented in Figure 5.

Although there are limited observations specific to the effects of insect mortality on water yield; Stednick and Jensen (2007) concluded that beetle induced mortality has the potential to change snow pack accumulation and melt processes, evapotranspiration, and subsequent stream flow. They also suggested that watersheds receiving less than 50.8 mm (20 in.) of precipitation would not demonstrate a change in water yield due to dieback.

Although most watershed experiments have looked at the effects of reducing forest cover on water yield, it has been demonstrated that any significant change in forest cover whether an increase or a decrease can cause a change in ET and result in a water yield modification. In contrast to the increases in water yield that have been observed to occur following reductions in forest cover, limited observations do indicate that increases in forest density can result in a proportional decline in water yield (Bosch and Hewlett 1982). In the North Platte River basin increases in forest cover have occurred naturally since the mid 1800’s on public lands and caused water yield from those lands to decline significantly over time (Leaf 1999; Troendle and Nankervis 2000; Troendle et al., 2003, 2006). As existing stands in the North Platte River basin matured, declines in water yield occurred. It is presumed that the increased forest cover that has occurred reflects both the re-growth following past timber harvesting activities as well as the expansion of forested area and increased densification of existing forest vegetation resulting from a reduced timber harvest program combined with a successful fire prevention program.
Figure 5. Documented responses following fire, insect mortality, and complete clear cutting compared with increases in water yield estimated from paired watershed studies (Troendle and Porth, in draft).

The primary objective of this study was to document, through field observations and simulation, the potential effect of insect mortality on water yield. A working hypothesis, based on what has been learned following long term watershed, plot, and process studies in the central and northern Rocky Mountains is that any form of forest disturbance that does not alter either the infiltration characteristics of the soil or the pathway that water takes to the stream channel would most likely result in changes in annual water yield. The latter changes are either similar to or parallel to those observed following timber harvest. In addition the increase in water yield would be proportional to the percent forest cover removed and the energy and precipitation available on site. The relationship expressed in Figure 5 represents a hypothesis, to be assessed in this study, on the effect of insect mortality on water yield in the North Platte River basin.

3. Field Protocol to Document Change

Although there are numerous alternatives to aid in documenting the hydrologic response to watershed disturbance, the ultimate proof of the effects of forest disturbance on water yield is measured at the stream gauge (Reynolds and Leyton, 1967; Hewlett et al., 1969). A study was designed as part of the Troendle et al. (2003) report on the Platte River. The study identified a series of gauged watersheds and supporting analytical data that would be useful in documenting future changes in water yield should catastrophic disturbance occur.
3.1. Site Selection

Troendle et al. (2003) identified a series of stream gauge, snow pack accumulation, and precipitation monitoring sites that would be useful as reference pairs for the future documentation of measurable long term changes in stream flow. Long term changes in flow were considered to be those that might occur as a result of future forest management activities as well as other land and water use changes. The intent was to identify existing long term stream gauges and well correlated long term snow course and/or precipitation records that could be used in double-mass and covariance analyses to document future change in water yield. The site selection process was described in detail by Porth in Appendix C of Troendle et al. (2003) and is summarized as follows:

The selection process was sequential in nature. First, criteria were established to screen and then select a series of stream gauges that had potential for use in documenting future changes in water yield. Because catastrophic impacts, such as insect or disease infestations, can be expected to cross watershed boundaries, the second step in the process was to identify stable climatic data that could be used as independent variables in the analyses to document changes in the dependent variable, stream flow. It was assumed that the climatic data would not be as subject to change as a result of the catastrophic impact and would provide a stable control in documenting change on the watershed of interest. It was anticipated that this assumption would need to be validated as part of any subsequent analysis and numerous controls were identified for each watershed selected. By pre-selecting watersheds suitable for change detection and identifying appropriate control variables for analysis the opportunity for post impact bias in the selection process is eliminated.

3.1.1. Watershed Selection

The study reported on by Troendle et al. (2003) was linked to the Platte River Recovery Program and as such 1997 was considered the reference year for stream gauge selection as well as the reference point for future analyses. In addition to being in current operation and having a stable record, each stream gauge considered had to have a minimum of 25 years of record prior to 1997. The rational for requiring at least 25 years of record was developed by Porth et al. (2001). In total, over 1800 stream gauges have existed in Colorado and Wyoming but only 27 operational stream gauges in Colorado and 20 in Wyoming were located on tributaries to either the North or South Platte River that met the initial selection criteria based on length of record. Daily stream flow data for the entire period of record was downloaded for each of the 47 stream gauging sites. Since some of the stream gauges are not maintained year around and because observed changes in flow due to vegetation disturbance have been demonstrated to occur only during the April to September period, only seasonal flows were of concern and downloaded. Seasonal flow is measured during the time period from approximately April 15 to September 30 each year.
Seasonal stream flow for the 47 Colorado and Wyoming gauging sites were then plotted in a double-mass comparison (Anderson 1955) with the stream flow for the Encampment River above Hog Park. Double-mass plots provide a very simple and comprehensive way to examine the relationship between two parameters and the technique is very sensitive to even the most subtle changes in the relationship within or between the variables. The Hog Park gauge had been demonstrated to be useful for such a screening process as it has a long record, the stream flow is not subject to diversion or augmentation, and there do not appear to be any detectable measurement issues (Troendle and Porth 2000). Based on this comparison, many of the 47 stream flow records did not appear suitable for future comparisons while others appeared quite good. The list of suitable stream gauging sites was reduced to 11 in Colorado and 8 in Wyoming as a result of this initial screening process (see Porth, Appendix C, Troendle et al., 2003).

3.1.2. Snow Courses

The next step in the process was to identify National Resources Conservation Service (NRCS) snow courses that would be useful in comparing snow pack accumulation (X) with stream flow (Y). Ten snow courses in Colorado and 10 courses in Wyoming with long term, overlapping records appeared to be located in or near the 19 watersheds of interest. Four of the Wyoming snow courses had been converted to Snotel (pillow) sites and were dropped from further consideration because the conversion from a manual site to an automated site alters the measurement. The snow course data from the 16 remaining manual snow courses was double-mass plotted over that for the Glade Creek snow course. The Glade Creek snow course had been identified in earlier studies to be stable over time and useful as a reference site (Troendle and Porth 2000). Four of the Colorado and 6 of the Wyoming snow courses were retained as being both stable and well correlated with the Glade Creek snow course (see Porth, Appendix C, Troendle et al., 2003).

3.1.3. Precipitation Gauges

In order to improve the design and analysis of the study, the availability of long-term precipitation records to compare with the stream flow data was also evaluated. Precipitation data for Colorado and Wyoming were obtained through the NRCS website. Ten representative sites, 8 in Colorado and 2 in Wyoming were located near the study watersheds, were in operation, and had the necessary years of continuous and overlapping record. Annual precipitation for those sites was then compared with the snow course data from Glade Creek in Wyoming using double-mass plots. All comparisons appeared to be stable and well correlated (see Porth, Appendix C, Troendle et al., 2003) so all 10 precipitation sites were retained for further analysis.
3.2. Assessment Procedure

In total, 19 stream gauges, 10 snow courses, and 10 precipitation gauges were identified and retained for further analysis (Troendle et al., 2003). In the final selection process, the stream flow records for the 19 stream gauges were double-mass plotted over the overlapping record for the 3 nearest snow courses and the 3 nearest precipitation gauges. Ten of the 19 stream gauges initially selected for consideration were not well correlated with a sufficient number of snow course and precipitation sites and were subsequently dropped from the analysis. Nine stream gauging sites were retained as reference sites as the stream flow from each of the watersheds exhibited a strong and stable relationship with data collected at up to 3 snow courses and 3 precipitation gauges. The cross comparisons between stream flow, snowpack accumulation, and annual precipitation as presented in Appendix C of Troendle et al. (2003) were quite exhaustive. The 9 watersheds selected in the North and South Platte River basin are depicted on Figure 6.

For each of the 9 reference watersheds identified on Figure 6, an analytical procedure was developed to assess future change in seasonal stream flow. The analytical procedure’s format 1) characterized the cumulative relationship between pre-1997 seasonal stream and an estimate of April 1 snow pack accumulation or annual precipitation measured at 3 or more nearby sites, 2) allows the addition of post-1997 data pairs to the cumulative relationship, and 3) plots and allows visual comparison of the pre-1997 stream flow relationship with the post-1997 observations. The expectation was that by plotting future data pairs on the pre-1997 relationship any demonstrated shifts that appear to have occurred in the various stream flow-precipitation relationships might warrant a covariance analysis to determine the statistical significance of what might appear to be a post-1997 departure in the pre-1997 relationship. The objective was to identify and pre-select reference watersheds that might be useful in documenting future changes in stream flow, should they occur. Identifying a matrix of control, or reference, sites for each stream gauge, rather than individual pairs, insures valid cross comparisons can be made that would document the change and verify the source of the change. A macro was developed to allow projection of the fitted line, with 95 percent prediction intervals, beyond 1997 and allow the addition of future (post-1997) data points. If any post-1997 point or points were to plot outside of the prediction intervals this would be a flag signifying the user must go back and perform a standard analysis of covariance on the original pre- and post-1997 data to determine if there has been a significant change in the watershed response since the reference year of 1997. An example of the double-mass plot of seasonal stream flow from the Encampment River above Hog Park plotted over peak water equivalent for the Glade Creek snow course is presented in Figure 7.
**Figure 6.** The 9 Platte River watersheds identified as reference watersheds to detect changes in streamflow resulting from landscape scale forest disturbance (map prepared by G. Walker Johnson, METI, Inc., unpublished).

The regression line and confidence bands plotted on Figure 7 represent the comparison of the seasonal stream flow from the Encampment River above Hog Park (Y) plotted over the peak water equivalent measured about April 1 at the Glade Creek snow course. The relationship between the data pairs is quite stable up through 1997. Data pairs for 1998 through 2011 have been added to the regression but were not used to influence the regression. It can be noted that in 2005 or 2006 the relationship between seasonal stream flow and snow pack peak water equivalent began to shift and that stream flow appeared to be increasing relative to measured peak water equivalent. Although not presented, the study design allowed 3 or more similar comparisons between seasonal stream flow and either peak water equivalent or annual precipitation. In the case of the Encampment River above Hog Park, all comparisons demonstrated a similar trend. In addition similar double-mass comparisons were made between all the combinations of snow course plotted over snow course, precipitation plotted over precipitation, and snow course plotted over precipitation to verify that it was the stream flow (Y) that was changing and not the independent variable (X).
Although not all watersheds demonstrated the dramatic shift that was evident for the Encampment River above Hog Park, all comparisons indicated that all the snow pack and precipitation measurements appeared stable and observed shifts were the result of a stream flow change. Much more detail on the double-mass analysis is presented in Troendle et al. (2003) and Troendle and Porth (2000).

Figure 7. Double-mass plot of seasonal streamflow from the Encampment River above Hog Park. Data was plotted over Peak Water Equivalent, about April 1, at Glade Creek snow course.

3.3. Assessment of Response

3.3.1. Water Yield

As part of a current study, the stream flow, snowpack, and precipitation records for the sites pre-selected by Troendle et al. in 2003 were brought up to date, the data pairs added to the regression models as shown in Figure 7, and shifts identified in the pre-defined relationships observed at the 9 gauging sites were evaluated. Unfortunately, the St. Vrain stream gauge and the Saratoga, WY
and the Longmont, CO precipitation gauges had to be deleted from further analysis as all three had either been moved or discontinued prior to 2011. For each of the 8 remaining reference watersheds, 5 to 7 comparisons were made between accumulated seasonal stream flow and either accumulated snowpack or annual precipitation. As noted above, if a shift or deflection in the double-mass relationship was observed for one comparison for a given watershed, that departure was observed for all comparisons for that watershed.

Of the 8 gauged watersheds available for analysis, only the Encampment River above Hog Park and North Brush Creek have no diversions and receive no imported water. The Bear Creek watershed, near Morrison, CO, has a small diversion to irrigate about 405 hectares (1000 acres) but is otherwise unregulated. The other 5 watersheds are regulated but the pre-1997 water yield from each of those watersheds was well correlated with the respective climatic input, thus were retained for further analysis. Because the double-mass plots indicated stream flow from at least some of the watersheds appeared to have increased in recent years, an Analysis of Covariance (ANCOVA) was conducted to test for changes in flow from all watersheds. For each of the 8 watersheds, ANCOVA analyses were conducted comparing stream flow data for each of the watersheds for pre- and post-infestation periods with the suite of annual precipitation estimates or snow course peak water equivalent data appropriate to the stream gauge. For the ANCOVA analyses 2002 was used as the reference year rather than 1997. The shift from 1997 to 2002 was necessary to accommodate the fact that the vegetation data to be used in subsequent analyses represents conditions that existed in 2002 and 2012. The data up to and including 2002 was considered pre-beetle and the data since 2002 considered post-beetle. Since 2002 data layer represents pre disturbance conditions, shifting the reference date does not present a problem while it adds 5 years to the length of pre disturbance measurements records, or calibration period.

An Analysis of Covariance (ANCOVA) was conducted between all possible pairs of stream flow (Y) and either snow pack (X\textsubscript{1}) or annual precipitation (X\textsubscript{2}). The ANCOVA analyses demonstrated that the pre- and post- 2002 estimate of adjusted mean seasonal stream flow for a given watershed was consistent for all 5 to 7 comparisons. However, estimated response varied by watershed in both the magnitude of adjusted mean seasonal stream flow as well as the estimate of the adjusted mean change in seasonal stream flow. ANCOVA results, based on a representative precipitation gauge and a representative standard snow course are presented for each watershed (Table 2). The Glade Creek snow course was selected as the representative snow course because it is the only site that is well correlated with the seasonal stream flow from all eight reference watersheds. As could be inferred from the double-mass plot comparisons, the ANCOVA’s indicated that seasonal water yield significantly increased on some watersheds while significant change could not be detected on others (Table 2). Bear Creek is unique in that the ANCOVA analysis suggests there may have been a decrease in water yield from that watershed (Table 2) in contrast to either an increase or no change in water yield on the other 7 watersheds. The decrease in stream flow from Bear Creek is significant in some comparisons, particularly those with annual precipitation rather than snow pack. Although not all comparisons are presented, they are well represented and encompassed by the two comparisons presented in Table 2.
It can be observed (Table 2) that both observed seasonal flow and the estimate of change in flow are quite variable between watersheds. The differences in the magnitude of seasonal flow between watersheds, and to some degree the magnitude of the change in flow, is primarily a function of differences in precipitation input. The differences in the estimate of the flow change are more a function of factors such as the combination of the vegetative mortality, species composition, initial stand conditions, precipitation input, and the timing of when mortality may have occurred during the 2002-2012 period.

Table 2.

ANCOVA analyses results between annual precipitation (P) and peak water equivalent in the snowpack (PWE) of the referenced watersheds

<table>
<thead>
<tr>
<th>Watershed (X)</th>
<th>Precipitation/ Snowpack Site (Y)</th>
<th>Pre-Disturbance Flow mm (in.)</th>
<th>Post-Disturbance Flow mm (in.)</th>
<th>Change in Flow mm (in.)</th>
<th>Probability of Significance</th>
</tr>
</thead>
<tbody>
<tr>
<td>North Platte River</td>
<td>Glade (PWE)</td>
<td>75.8 (3.0)</td>
<td>102.5 (4.0)</td>
<td>26.6 (1.0)</td>
<td>0.06</td>
</tr>
<tr>
<td>North Platte River</td>
<td>Grand Lake (P)</td>
<td>78.7 (3.1)</td>
<td>105.2 (4.1)</td>
<td>26.5 (1.0)</td>
<td>0.07</td>
</tr>
<tr>
<td>North Brush Creek</td>
<td>Glade (PWE)</td>
<td>393.2 (15.5)</td>
<td>528.4 (20.8)</td>
<td>135.2 (5.3)</td>
<td>0.02</td>
</tr>
<tr>
<td>North Brush Creek</td>
<td>Grand Lake (P)</td>
<td>392.5 (15.4)</td>
<td>531.4 (20.9)</td>
<td>138.9 (5.5)</td>
<td>0.01</td>
</tr>
<tr>
<td>Encampment River</td>
<td>Glade (PWE)</td>
<td>457.1 (18.0)</td>
<td>608.8 (24.0)</td>
<td>151.6 (6.0)</td>
<td>0.01</td>
</tr>
<tr>
<td>Encampment River</td>
<td>Grand Lake (P)</td>
<td>460.4 (18.13)</td>
<td>595.0 (23.42)</td>
<td>134.6 (5.3)</td>
<td>0.02</td>
</tr>
<tr>
<td>Encampment River</td>
<td>Glade (PWE)</td>
<td>261.3 (10.3)</td>
<td>339.5 (13.4)</td>
<td>78.2 (3.1)</td>
<td>0.01</td>
</tr>
<tr>
<td>Encampment River</td>
<td>Grand Lake (P)</td>
<td>260.6 (10.3)</td>
<td>334.7 (13.2)</td>
<td>74.1 (2.9)</td>
<td>0.04</td>
</tr>
<tr>
<td>Laramie River</td>
<td>Glade (PWE)</td>
<td>115.2 (4.5)</td>
<td>140.1 (5.5)</td>
<td>24.8 (1.0)</td>
<td>0.19</td>
</tr>
<tr>
<td>Laramie River</td>
<td>Grand Lake (P)</td>
<td>115.5 (4.5)</td>
<td>145.2 (5.7)</td>
<td>29.7 (1.2)</td>
<td>0.07</td>
</tr>
<tr>
<td>Bear Creek</td>
<td>Glade (PWE)</td>
<td>82.5 (3.2)</td>
<td>58.1 (2.3)</td>
<td>-24.4 (-1.0)</td>
<td>0.19</td>
</tr>
<tr>
<td>Bear Creek</td>
<td>Boulder (P)</td>
<td>81.7 (3.2)</td>
<td>48.6 (1.9)</td>
<td>-33.1 (-1.3)</td>
<td>0.04</td>
</tr>
<tr>
<td>Big Thompson River</td>
<td>Glade (PWE)</td>
<td>284.6 (11.2)</td>
<td>284.0 (11.2)</td>
<td>-0.6 (-0.0)</td>
<td>0.98</td>
</tr>
<tr>
<td>Big Thompson River</td>
<td>Grand Lake (P)</td>
<td>281.4 (11.1)</td>
<td>289.3 (11.4)</td>
<td>7.9 (0.3)</td>
<td>0.76</td>
</tr>
<tr>
<td>Cache La Poudre River</td>
<td>Glade (PWE)</td>
<td>97.8 (3.9)</td>
<td>97.0 (3.8)</td>
<td>-0.7 (-0.0)</td>
<td>0.95</td>
</tr>
<tr>
<td>Cache La Poudre River</td>
<td>Waterdale (P)</td>
<td>96.0 (3.8)</td>
<td>92.2 (3.6)</td>
<td>-3.8 (-0.1)</td>
<td>0.78</td>
</tr>
</tbody>
</table>
At this point in time, a significant change in stream flow from the Big Thompson River and the Poudre River watersheds cannot be detected with the data available. The P values for the changes in flow in the Laramie River watershed and from the North Platte River above Northgate, as shown in Table 2, are suggestive that change may have occurred. The two Encampment River watersheds and the North Brush Creek watershed demonstrated that a significant increase in stream flow has occurred for all the comparisons made between seasonal stream flow and either snowpack accumulation or annual precipitation measures. Bear Creek is the only watershed consistently expressing a decrease in seasonal flow that is significant when related to annual precipitation.

Only two of the 5 to 7 ANCOVA analyses conducted for each of the watersheds are depicted in Table 2. In general, snow pack accumulation appeared to be the better predictor of seasonal flow and change in water yield for the North Platte watersheds, while annual precipitation was the better predictor in the South Platte watersheds. The precipitation amount and distribution differs between the North and South Platte River watersheds (Troendle et al., 2003). Less precipitation occurs in the South Platte watershed while a greater percentage of the annual precipitation occurs in the summer, thus the greater dependence of stream flow on annual precipitation rather than winter snow pack. More precipitation falls on the North Platte watershed and seasonal stream flow is dominated by the melting of the winter snow pack.

### 3.3.2. Vegetation

The U. S. Forest Service, Forest Health Technology Enterprise Team (FHTET) located in Fort Collins, Colorado provided two raster based vegetation layers describing vegetative conditions across the study area for the years 2002 and 2012. Using Geographic Information Systems (GIS), the two data layers were clipped to match the watershed boundary for each of the 8 study watersheds as well as for the entire North Platte watershed above Seminoe Reservoir in Wyoming. In contrast to what was available for the earlier studies on the Platte River (Troendle and Nankervis 2000; Troendle et al., 2003; Troendle et al., 2006), the estimates of vegetative cover provided by FHTET apply to all forest land in the watershed, not just Forest Service land. In addition, the estimate of the changes that occurred between 2002 and 2012 include any increase or decrease in basal area resulting from in-growth, re-growth, or mortality resulting from fire, insects and disease, or timber harvest. The earlier studies dealt only with vegetation on National Forest lands as estimated using the single R2VEG data layer in a polygon format, also provided by the US Forest Service (see Troendle and Nankervis 2000). Ellenwood, et al. (in press) provided the following abstract describing the development of the two vegetation data layers made available for this study in 240m X 240m raster format:
“Parameter datasets of presence, basal area (BA), and stand density index (SDI) were modeled for 289 individual tree species using forest inventory data and several national datasets for the contiguous US and Alaska at a 30-meter resolution. Existing national datasets utilized include: the National Land Cover Database (NLCD) project which created a 30-meter three-season Landsat dataset for each of 76 USGS map zones; the USDA NRCS localized SSURGO and regionalized STATSGO soil datasets; the USDA Forest Service Forest Inventory and Analysis (FIA) nationwide forest inventory collected between 1997 and 2009; NOAA National Climate Data Center climate normal (1971 to 2000) data for 7937 weather stations (CLIM81), and the 2004 USGS National Elevation Dataset (NED) at 30 meter ground resolution.

Predictor variables were generated from the three-season Landsat 7 imagery, NRCS-derived soils data, NED-derived terrain data, and CLIM81-derived climate data. Models of individual tree species parameters were generated by linking the FIA inventory data to the predictor data layers and analyzed with See5/Cubist data mining software. Model outputs were applied to the 30-meter spatial datasets using ERDAS Imagine software. For each tree species in a mapping zone: presence, basal area, and stand density index models were created for a total of 8,402 models.

To create better models, data selection and a number of techniques were incorporated to improve model representation. FIA evaluation groups (formerly known as cycles) were selected to correspond with imagery dates with 80 percent of the FIA plots being sampled within 5 years of the imagery collection. The mismatch between the inventory subplot size of 1/24th acre (168.6 m²) and the 30-meter spatial dataset resolution (900 m²) were accounted for by using a Poisson distribution adjustment to the plot-level summaries. Cubist modeling biases due to optimization of model fits to median values were adjusted by matching the geospatial product histogram to the inventory histogram. Additional adjustments were made to balance the sum of the individual species parameter to the overall total parameter.

With image collection dates ranging from 1985 to 2005 and a mean collection date for tree areas of 2002, the vintage of the resultant parameter datasets are considered as 2002. An additional adjustment was applied to account for growth and mortality to bring the current dataset to the 2010/2012 vintage. The adjustment process for the contiguous US utilized a MODIS phenology dataset from NASA-Stennis (NASA, 2001–2010) and the Forest Health Protection Mapping and Reporting portal Forest Disturbance Mapper (FDM, 2008-2012). The NASA dataset was utilized for the entire contiguous US and the FDM dataset was utilized in key portions of Colorado, Utah, and Wyoming to account for recent fire and mountain pine beetle mortality. A linear regression was performed on the 10-year stack of the annual 80 percent Normalized Difference Vegetation Index (NDVI) layer for the years ranging from 2001 to 2010. Geospatial products of the phenology regression slope and regression r-square were created. Re-measured FIA sub-plots (~170,000) were utilized to scale the phenology regression layers. Annual BA change (including both growth and mortality) was determined by subtracting current-period BA from the previous-period BA and dividing the result by the period between the plot measurements. This scaling was weighted by inventory age to create a percentage change product and applied to the complete 2002 vintage dataset to create an updated 2010/2012 vintage dataset to support the modeling of the National Insect and Disease Risk Map.”
Most of the FIA data, as well as other available inventory data, were used in training the imagery and developing the data layers described above leaving little opportunity to evaluate or QA/QC the layers, once developed. As part of the initial QA/QC protocol, the FIA data used in the construction of the layers was also used to verify that the data was properly portrayed in the final product. This was not intended to represent a true validation. Recently collected FIA data, not used in the development of the layers, is being used to validate the layers. Although the comparisons are quite reasonable, mortality appears to have increased relative to initial (2012) estimates (personal communication, J. Ellenwood, USFS, FHTET, Fort Collins, CO).

The vegetation layers provided by FHTET were clipped using the study watershed boundaries depicted in Figure 7. The 240 X 240 m pixel data was summarized by watershed using GIS to determine the watershed area, forested area, percent of area forested, and change in basal area, by watershed, for the 2002 to 2012 period (Table 3). The changes in basal area that occurred across the entire North and South Platte River study are depicted in Figure 8. Decreases in basal area are in red while increases in basal area are depicted in a bright green. The colors are scaled to represent magnitude of change but the scale of the map is such that those differences are not discernible. The decrease in basal area, inferred to be the result of insect mortality, is quite extensive. Interestingly, increases in basal area appear to be significant in some areas, particularly in portions of the South Platte watershed.
Figure 8. Basal area changes during 2002 to 2012 in the study area. Red implies reduction in basal area within the watershed boundary and brighter green implies increase in basal area. Although there is a color gradient, it is not apparent at this map scale (map prepared by G. Walker Johnson, METI, Inc., unpublished).
Table 3.
Summary characteristics of the 8 study watersheds between 2002 and 2012

<table>
<thead>
<tr>
<th>Watershed</th>
<th>Watershed Area km² (mi²)</th>
<th>Forested Area km² (mi²)</th>
<th>Percent Area Forested</th>
<th>Percent Change in Basal Area</th>
</tr>
</thead>
<tbody>
<tr>
<td>North Platte River</td>
<td>3,703 (1430.0)</td>
<td>1,668 (644.0)</td>
<td>45</td>
<td>-48.7</td>
</tr>
<tr>
<td>North Brush Creek</td>
<td>97 (37.5)</td>
<td>97 (37.5)</td>
<td>100</td>
<td>-37.0</td>
</tr>
<tr>
<td>Encampment River above Hog Park</td>
<td>189 (72.8)</td>
<td>180 (72.1)</td>
<td>99</td>
<td>-66.5</td>
</tr>
<tr>
<td>Encampment River at Mouth</td>
<td>678 (261.8)</td>
<td>502 (193.7)</td>
<td>74</td>
<td>-59.6</td>
</tr>
<tr>
<td>Laramie River</td>
<td>1,116 (430.9)</td>
<td>815 (314.6)</td>
<td>73</td>
<td>-40.7</td>
</tr>
<tr>
<td>Bear Creek</td>
<td>426 (164.5)</td>
<td>388 (149.7)</td>
<td>91</td>
<td>+5.6</td>
</tr>
<tr>
<td>Big Thompson River</td>
<td>357 (138.0)</td>
<td>257 (99.4)</td>
<td>72</td>
<td>-14.4</td>
</tr>
<tr>
<td>Cache La Poudre River</td>
<td>2,735 (1056.0)</td>
<td>1,969 (760.3)</td>
<td>72</td>
<td>-27.8</td>
</tr>
</tbody>
</table>

The estimated changes in water yield as presented in Table 2 represent the change in flow measured at the stream gauge and as such represents the equivalent of a layer of water uniformly produced from the entire watershed area. However, the assumption is being made in this study that the change in water yield, measured at the stream gauge, is the effect of the reduction in basal area that occurred on the forested portion of the watershed. Unfortunately, the watersheds are not fully forested and comparison of the flow change estimated for the entire watershed with the basal area change on the forested portion requires that either the estimate of reduction in basal area on the forested portion of the watershed or the change in flow estimated for the entire watershed area be transformed or scaled, so the numbers are comparable. To accomplish this, the estimated change in seasonal water yield for the entire watershed as presented in Table 2 was adjusted to equal the change in flow that would have had to occur from the forested portion of the watershed. The adjustment, or scaling, was done by dividing the Change in Flow from Table 2 by the Percent Area Forested from Table 3 and then multiplying by 100. This calculation prorates the change in flow estimated to have occurred over the entire watershed (from Table 2) to an equivalent depth of water that would have had to be produced if the entire change in flow came from only the forested portion of the watershed (Table 4). The adjusted changes in water yield, estimated from the ANCOVA analysis with the Glade Creek snow course, were compared with the changes in basal area estimated to have occurred during the 2002 to 2012 time period for each of the 8 watersheds (Figure 9).
Table 4.
Estimated changes in basal area and flow adjusted to the forested portion of the study watersheds for 2002-2012

<table>
<thead>
<tr>
<th>Watershed</th>
<th>Precipitation/Snowpack Site</th>
<th>Adjusted Change in Flow (mm (in.))</th>
<th>Change In Basal Area (Percent of Forested Area)</th>
</tr>
</thead>
<tbody>
<tr>
<td>North Platte River</td>
<td>Glade (PWE)</td>
<td>58.9 (2.3)</td>
<td>-48.7</td>
</tr>
<tr>
<td>North Platte River</td>
<td>Grand Lake (P)</td>
<td>58.6 (2.3)</td>
<td>-48.7</td>
</tr>
<tr>
<td>North Brush Creek</td>
<td>Glade (PWE)</td>
<td>135.4 (5.3)</td>
<td>-37.0</td>
</tr>
<tr>
<td>North Brush Creek</td>
<td>Grand Lake (P)</td>
<td>147.7 (5.8)</td>
<td>-37.0</td>
</tr>
<tr>
<td>Encampment River above Hog Park</td>
<td>Glade (PWE)</td>
<td>153.4 (6.0)</td>
<td>-66.5</td>
</tr>
<tr>
<td>Encampment River above Hog Park</td>
<td>Grand Lake (P)</td>
<td>136.1 (5.4)</td>
<td>-66.5</td>
</tr>
<tr>
<td>Encampment River at Mouth</td>
<td>Glade (PWE)</td>
<td>105.4 (4.1)</td>
<td>-59.6</td>
</tr>
<tr>
<td>Encampment River at Mouth</td>
<td>Grand Lake (P)</td>
<td>99.9 (3.9)</td>
<td>-59.6</td>
</tr>
<tr>
<td>Laramie River</td>
<td>Glade (PWE)</td>
<td>33.9 (1.3)</td>
<td>-40.7</td>
</tr>
<tr>
<td>Laramie River</td>
<td>Grand Lake (P)</td>
<td>40.6 (1.6)</td>
<td>-40.7</td>
</tr>
<tr>
<td>Bear Creek</td>
<td>Glade (PWE)</td>
<td>-26.7 (-1.0)</td>
<td>+5.6</td>
</tr>
<tr>
<td>Bear Creek</td>
<td>Boulder (P)</td>
<td>-36.2 (-1.4)</td>
<td>+5.6</td>
</tr>
<tr>
<td>Big Thompson River</td>
<td>Glade (PWE)</td>
<td>24.2 (1.0)</td>
<td>-14.4</td>
</tr>
<tr>
<td>Big Thompson River</td>
<td>Grand Lake (P)</td>
<td>18.8 (0.7)</td>
<td>-14.4</td>
</tr>
<tr>
<td>Cache La Poudre River</td>
<td>Glade (PWE)</td>
<td>0.0 (0.0)</td>
<td>-27.8</td>
</tr>
<tr>
<td>Cache La Poudre River</td>
<td>Waterdale (P)</td>
<td>-5.4 (-0.2)</td>
<td>-27.8</td>
</tr>
</tbody>
</table>

The basal area reductions on those watersheds where a significant change in water yield has been detected (North Brush Creek, Encampment River above Hog Park, and Encampment River at the Mouth) are above the threshold of 20 percent considered to be the minimum necessary to result in a significant increase in water yield (Bosch and Hewlet 1982; Stednick 1996). Cache La Poudre River did not exhibit a flow change, yet the estimated decrease in basal area exceeds the threshold needed to cause a change.
Basal area reduction on North Brush Creek appears low given the large increase in flow. Bear Creek is again unique in that it demonstrated both an increase in vegetation density and a decrease in flow, a reasonable but somewhat unexpected response.

In general, the regression between change in water yield and the percent reduction in cover, or basal area, is quite good (Figure 9). The $R^2$ of 0.65 is less than the $R^2$ of 0.77 determined for the paired watershed study responses presented in Figure 3, but the slopes of the two regressions are similar. Two outliers, North Brush Creek (high) and Cache La Poudre River (low), are the primary cause of the reduced $R^2$. Earlier, it was noted that Stednick (1996) determined that the average increase in water yield observed from the 95 paired watershed experiments conducted nationally was 2.5 mm (0.1 in.) per 1 percent reduction in forest cover. The paired watershed studies conducted in the Rocky Mountain region, as presented in Figure 3, demonstrated a 2.33 mm (0.09 in.) increase in water yield per 1 percent reduction in forest cover. The response to insect mortality, as presented in Figure 8, represents a 2.22 mm (0.09 in.) increase in water yield per 1 percent reduction in forest cover. The two regressions differ by approximately 5 percent.

![Figure 9. Regression relationships between observed changes in water yield at the 8 USGS stream gauges. Changes were plotted over percent reduction in cover on the forested portion of the watershed. The two points that fall outside the confidence band are North Brush Creek (high) and Cache La Poudre River (low).](image-url)
Based on what we have learned from the paired watershed studies focused on timber harvest, watershed response to insect mortality appears to parallel observations following timber harvest (Figure 10). The adjusted stream flow increase as a function of peak water equivalent (PWE) at Glade Creek compares quite well with the regression relationship between observed increase in stream flow and decrease in forest cover following timber harvest on the experimental watersheds initially presented as Figures 3 and 5 and reproduced again in Figure 10.

![Graph](image)

**Figure 10.** Estimates of change in average seasonal flow and forested watershed area. The data points represent estimates of the change in average seasonal flow from Table 4 (ANCOVA analysis with Glade Creek PWE) plotted over the estimate of the reduction in cover (basal area) that occurred during the period 2002-2012. The regression line and confidence bands represent that for the paired watershed studies (Figures 3 and 5) the regression line has been extended below 0.0, to allow comparison with the decrease in water yield observed for Bear Creek.

As one would expect, there is variability in the estimates of the increase in flow from the study watersheds with North Brush Creek being the only observation that falls above the regression line for the paired watershed responses (Figure 10). All other points plot below the regression line, with both the Cache La Poudre River and Laramie River plotting just below the 95 percent confidence band about the mean of the regression line. Overall, the results of the analysis of the vegetative changes and observed stream flow response from the 8 study watersheds appear to parallel, and perhaps equal, the responses documented in paired watersheds studies for similar degrees of disturbance. The significant increase in flow on North Brush Creek appears to be high relative to the estimated basal area change on the watershed. In contrast, the flow change from Cache La Poudre River watershed appears to be low given the estimated basal area reduction.
North Brush Creek channel was severely scoured in 2010 and visual observation indicated modification of the channel has occurred requiring that a new cross section and rating curve be established (personal communication Greg Bevenger, USFS Regional Hydrologist, Region 4 and Kirk Miller, USGS Cheyenne) and that may partially explain the discrepancy. Although the USGS gauging protocol would indicate the same measurement resolution is being met following scour, the estimate of discharge might still have been altered relative to pre-scour conditions. The observation that the flow change in Cache la Poudre watershed seems low relative to the magnitude of basal area reduction cannot be explained at this point. Cache la Poudre River is heavily regulated and we can only assume there have not been any changes in the operational hydrology of the river that would offset any change in flow.

4. Hydrologic Simulation

4.1. The WRENSS Hydrologic Model

The WRENSS Hydrologic model (Troendle 1979; Troendle and Leaf 1980; Bernier 1986; Swanson 1994; Troendle and Nankervis 2000; Troendle et al., 2003) was used to simulate the changes in water available for stream flow from the study watersheds as well as the entire watershed area in the North Platte River above Seminoe Reservoir. WRENSS is an acronym for the Environmental Protection Agency (EPA) funded handbook prepared by the U. S. Forest Service, titled Water Resource Evaluation of Non-point Silvicultural Sources (US EPA 1980). The WRENSS Hydrologic model was developed by Troendle and Leaf (1980) and presented as Chapter III in the WRENSS handbook. It was modified in the late 1980’s as referenced above, and was used in this study to simulate the water yield changes that occurred on the 8 study watersheds as well as for the entire North Platte River watershed contributing to the Seminoe Reservoir in Wyoming. Comparing the observed flow changes from the 8 study watersheds with the simulated flow changes allows assessment of the performance of the WRENSS hydrologic model and validates the appropriateness of using the WRENSS model in simulating flow change where flow change cannot be documented such as above the Seminoe Reservoir.

As noted, the WRENSS hydrologic model was initially developed in the late 1970’s as a tool useful in predicting the hydrologic effects of forest disturbance, particularly timber harvest, on annual water available for stream flow. The emphasis in the WRENSS hydrologic model is to simulate the change in water available for stream flow that occurs following disturbance, with less emphasis placed on simulating or distributing total stream flow. By definition, the WRENSS hydrologic model does not simulate changes in stream flow per se; it simulates changes in water available for stream flow. The distinction is that there is no routing component in WRENSS, although it is assumed that the changes in water available for stream flow are synonymous with stream flow change. There are empirical functions attached to the WRENSS model that distribute the simulated changes in water available for stream flow over an “average” hydrograph or flow frequency curve but they were not implemented in this effort. As part of the original WRENSS handbook development, the United States was partitioned into 7 hydrologic units or areas of similar hydrologic response (Figure 11).
This was done as a means of partitioning the variability in stream flow and watershed response to improve the prediction accuracy following forest disturbance. Hydrologic unit boundaries were based on Bailey’s Ecoregion Classification System (Bailey 1974), which like the hydrologic cycle is driven by precipitation and energy. The hydrologic boundaries for WRENSS have since been modified, by Bailey, to be consistent with the more recent definition of Ecoregion province boundaries (Bailey 1994).

![Map of Hydrologic regions and provinces of the conterminous US](image)

**Figure 11.** WRENSS Hydrologic Region boundaries (Troendle and Leaf 1980) overlaid with the most recent (Bailey 1994) Eco-region map (map prepared by R. Bailey, unpublished).

In developing the WRENSS hydrologic model, appropriate deterministic or process based models were first calibrated to observed experimental watershed responses, by region. The emphasis in model calibration was to simulate the flow changes following disturbance on those watersheds more so than simulating total stream flow. Because observations on watershed response to disturbance were limited to a relatively small number of experimental watersheds, the calibrated models were then used to develop empirical coefficients to adjust the water balance over a full range of disturbance activities. Initially in the WRENSS Handbook development process it was the intent to provide both the deterministic models as well as the empirical relationships for use in the watershed assessment process.
Unfortunately, it quickly became apparent that the necessary data needed to run deterministic models in management scenarios at landscape scales would not be realistic because of the lack of necessary and critical data. WRENSS was therefore developed as a nomographic procedure utilizing regionally derived empirical relationships to characterize the effects of forest disturbance on water available for stream flow. In the process of developing the empiricisms using deterministic models, most attention was given to defining the most critical components of the water balance affected by forest disturbance by hydrologic region and developing empiricisms to express them and assess their response in terms of metrics that would be available to managers. Thus the statement that WRENSS was developed to focus on change prediction rather than the absolute although WRENSS has been shown to simulate total water yield reasonably well (Troendle et al., 2007). In application, the WRENSS hydrologic model can be run at any scale. The only requirement is that the inputs related to energy, precipitation, and vegetation is at a common resolution, or scale, for each unit being simulated. The simulated responses from multiple units can be aggregated to estimate the total response for more extensive areas. Since all calculations represent a seasonal or annual response, routing from unit to unit is not an issue.

The regionalization of WRENSS was based on a combination of hydrologic and ecological considerations. Although the general approach to modeling is similar across all 7 hydrologic regions, the estimation of potential evapotranspiration (ET\text{pot}), actual evapotranspiration (ET\text{act}), and the coefficients used to modify those estimates to reflect the effect of forest condition differ by region. Differences in precipitation (amount, seasonal distribution, and form), energy (aspect, elevation), and tree species density and composition (expressed as leaf area index, basal area, or cover density) were the primary basis for regionalization as they represent the most significant factors affecting hydrologic response (Troendle and Leaf 1980). As in earlier studies of the Platte River, a modified procedure for the Rocky Mountain Region (Hydrologic Region 4, Figure 11) was used in this effort (See Troendle et al., 2003, 2006, 2007, 2010 for details).

In addition to identifying the appropriate Hydrologic Region, the site specific information needed to run the WRENSS hydrologic model includes estimates of monthly precipitation, aspect, species composition and density (i.e. leaf area index, basal area, or cover density), and the forest cover at which full hydrologic utilization of the site occurs must be input. This information is needed for each unit simulated, however, like units can be aggregated for simulation. The procedures for some hydrologic regions utilize additional information such as relative soil depth as an index to soil water availability where soil water storage may be a factor in the water balance. Others, such as the Rocky Mountain Region, require an estimate of the height of surface roughness which is necessary in estimating the retention, sublimation, and scour of the winter snowpack.

The concept of complete hydrologic utilization (BA\text{max}, LAI\text{max}, or CD\text{max}) implies that there is a point in the evolution of a stand when the forest cover becomes capable of using all the water and energy available on that site. Cover in excess of the level of complete hydrologic utilization might be capable of using more water but the water is not usually available for use except in wetter years. Therefore, the stand is under stress much of the time and more subject to mortality from a variety of stress related factors.
Less cover than required for complete hydrologic utilization would imply that on average some of the water available on site is excess and would be available for either stream flow increases or consumptive use by the residual vegetation. The initial simulation of water available for stream flow, using the WRENSS model, includes estimating the ET that would occur on site if the site were fully occupied by vegetation and fully utilizing the energy and precipitation available and at the point of complete hydrologic utilization of the site. The simulation of complete hydrologic utilization for the site may or may not be reflective of the vegetation conditions actually present on the site or the ET occurring from the site prior to any subsequent disturbance. A second simulation is necessary to adjust the initial simulation to reflect the vegetative cover conditions that actually exist on-site prior to disturbance. Once done, a third simulation adjusts the existing vegetative condition to simulate effect of disturbance or management activity on water available for stream flow. As noted, the focus of the simulation process is to characterize the potential changes in stream flow that occur, and is estimated as the change in water yield that occurs between successive simulations. With the exception of the function for estimating canopy interception loss from the winter snowfall in the Rocky Mountain region, the ET modifier coefficients developed for WRENSS are not linearly related to reduction in cover. The coefficients are sensitive to and reflect the increase in water use by residual vegetation following cover reduction as well as the increased sublimation losses that occur as the energy load and wind exposure increases near the forest floor following canopy removal. Data suggest the reduction of the canopy interception losses following tree removal is linear function of the reduction in cover.

An example of one of the empiricisms used in the WRENSS hydrologic model is the characterization of the interception and vaporization of snow in the forest canopy for the Rocky Mountain region, Hydrologic Region 4. Reductions in winter interception losses from the forest canopy have been shown to be a linear function of reductions in leaf area index or basal area (Wilm and Dunford 1948; Goodell 1952; Love 1953; Packer 1962; Gary and Troendle 1982; Potts 1984; Gary and Watkins 1985; Meiman 1987; Troendle 1987a; Troendle et al., 2003, 2006, 2010). The empirical relationship currently used in the Rocky Mountain Modified procedure is shown in Figure 12. The relationship presented has evolved since WRENSS was first developed as the result on the myriad of studies conducted since the late 1970’s involving partial tree removal and measured snow pack accumulation. Over 500 paired plot and watershed years of data are represented by the relationships presented in Figure 12. To some extent, aspect is a surrogate for species composition. On north slopes, the dominant tree species was mostly mature spruce-fir with a lodgepole pine cohort. On east, west, and south slopes the dominant species was mature lodgepole pine with a spruce-fir cohort. When characterizing the potential effect of canopy disturbance on snow pack accumulation, it is a prerequisite that BA_{max} for the site be known as well as the basal area that existed prior to the disturbance in the event the initial stand is not at complete hydrologic utilization. The relationship presented in Figure 12 can be applied using basal area directly but it must be remembered that in comparing two sites, any initial difference in basal area between the sites would need to be considered prior to assessing change following disturbance to one of the sites. The estimates of winter interception loss based on relationships presented in Figure 12 are quite consistent with historical estimates of interception loss (Kittredge 1949; Coleman 1953) but were developed from the controlled plot and watershed studies cited.
It is assumed that 100 percent of the gross precipitation to a site is estimated by that which falls into a clearing (forest clearing) or is measured in a rain gauge. As forest cover increases from 0.0 toward the point of complete hydrologic utilization, or BA\textsubscript{max}, the percent of gross precipitation that reaches the ground under the forest canopy decreases. The data indicated little or no further decrease was detectable once the cover reached 90 percent of BA\textsubscript{max}. In order to use Figure 12, an estimate of BA\textsubscript{max} for the site, an estimate of the current basal area on the site (assuming it is equal to or less than BA\textsubscript{max}), and the site aspect must be known. Divide the existing basal area by BA\textsubscript{max} and multiply the result by 100. Enter that percentage on Figure 12 to calculate the percentage of gross precipitation that should accumulate on the forest floor (through fall). Multiply the percentage of through fall by the estimate of gross precipitation to estimate the water equivalent that should be in the snow pack. This is a powerful relationship in that it allows comparison of what might be the expected water equivalent in the snow pack under the canopy of virtually any two stands, prior to disturbance, assuming a valid estimate of gross precipitation is available for each site. This estimate can be used in a post-disturbance scenario to estimate potential change when pre-impact measurements were not available. This would be preferable to simply comparing differences between two stands, after impact, and considering difference to be an estimate of change.

**Figure 12.** Percentage of the gross winter precipitation reaching the forest floor in the Central and Northern Rocky mountains as stand density increases from 0 (opening) to 100-percent of the BA\textsubscript{max} for the site (Troendle et al., 2010).
The ET modifier coefficients that represent the remaining components of the water balance are more complex, non-linear, and also a function of the aspect of the site and in application are integrated as a function of BA_{max}. It should be noted that if the initial forest cover exceeds that required for complete hydrologic utilization (i.e. BA_{max}) alterations in cover will not result in a change in water yield until the cover is reduced below BA_{max}. This would imply that the thinning of dense dog hair stands of lodgepole pine or overstory removal in old, mature stands might not produce an increase in stream flow.

In this study, the vegetation descriptions were derived from the 240 X 240 meter resolution data layers described earlier. For each pixel, the species composition and basal area for the 2002 layer was considered to reflect pre-beetle conditions and the 2012 vegetation layer represented post-beetle conditions. Because of the sheer number of pixels representing each watershed the resolution of the simulation was maintained at the level of the pixel but only the dominant specie on the pixel was selected to represent that pixel as a means of reducing the sheer number of simulations. To accomplish this, the basal areas on each pixel were combined into four species groups: 1) spruce-fir was the sum of the Engelmann spruce, subalpine fir, and blue spruce; 2) the Ponderosa pine group was the sum of Ponderosa pine and Douglas fir; 3) lodgepole pine; and 4) aspen. There were other species present but they were very trivial and not considered. The total basal area for the pixel was estimated as the sum of the basal areas for all species present but the pixel was considered to be represented by the dominant specie present in the simulations. This procedure introduced the possibility that some species may have been under or over represented in the simulation process but the consequences are not considered significant. Visual inspection of the pixel data indicated that the specie selected to represent the pixel was usually by far the dominant specie. Also, since each watershed simulation involved thousands of pixels and all four aspects there is little opportunity for bias towards one species or another in the simulation process. Once the dominant specie was identified, it was then necessary to estimate, for each of the four tree species groups, the basal area that reflected the point of “complete hydrologic utilization”, or basal area maximum (BA_{max}) for the site.

Two approaches to defining BA_{max} using the vegetation layers were assessed. The primary approach to defining the point of “complete hydrologic utilization” using the raster basal area layers was to plot the distribution of the number of pixels in each of the eight watersheds over the basal area represented on the pixel for each of the specie groups. The basal area representing the average modal point in the distribution was selected as the point of complete hydrologic utilization. An example of the distributions of basal area, by watershed, for lodgepole pine is depicted in Figure 13. Based on the relative uniformity of the distributions for all 8 watersheds, a single basal area estimate of 140 ft.² per acre was selected for all lodgepole pine as the point of complete hydrologic utilization. Following similar evaluations, a basal area of 150 ft.² per acre was selected for the spruce-fir group, 110 ft.² per acre for aspen, and 90 ft.² per acre for the ponderosa pine group. In the 2003 and 2006 studies of the Platte River Basin (Troendle et al., 2003, 2006) the R2VEG polygon data was used in the assessment and a BA_{max} of 120 ft.² per acre for lodgepole pine, 140 ft.² per acre for spruce-fir, 80 –110 ft.² per acre for the ponderosa pine group, and 110 ft.² per acre for aspen were estimated as the points of complete hydrologic utilization for use in the simulations associated with those studies.
Although the two estimates of $BA_{\text{max}}$ were derived using different databases describing the vegetation cover (a raster data set derived from spatial imagery vs. a polygon data set derived from field surveys and type mapping), the estimates of $BA_{\text{max}}$ are quite comparable given expected sampling variability and the finer resolution of the more robust raster data set.

To verify that the selection of $BA_{\text{max}}$ as described above was reasonable and to assess the sensitivity of the WRENSS model to the selection of $BA_{\text{max}}$, a second approach was implemented to define the point of complete hydrologic utilization for each species and used in comparative simulations. A frequency curve (the cumulative percentage of all pixels plotted over basal area) was constructed for each species and the basal area value at the 95 percentile was selected as the point of complete hydrologic utilization. This implies that 95 percent of all pixels have a basal area less than the selected value. The 95 percentile can be considered synonymous with maximum stand density. The 95 percentile also represents a level of cover well beyond the point of full hydrologic utilization. Using this approach, estimates of $BA_{\text{max}}$ of 191 ft.$^2$ per acre for lodgepole pine, 224 ft.$^2$ per acre for spruce-fir, 101 ft.$^2$ per acre for aspen, and 98 ft.$^2$ per acre for ponderosa pine was selected for use in the comparative simulations. These latter values are much higher than either the estimates derived using the estimate based on the modal point in the pixel distribution as described above or the values derived from the R2VEG polygon vegetation data used in earlier studies of the Platte River.
Simulations of the hydrologic response to the vegetation changes estimated to have occurred between 2002 and 2012 on the eight study watersheds, as well as the entire North Platte basin above Seminole Reservoir were completed for both estimates of BA\textsubscript{max}, or the points of “complete hydrologic utilization”, although only the simulated results using the modal estimate of BA\textsubscript{max} will be presented in detail.

In a procedure similar to the earlier studies (Troendle and Nankervis 2000; Troendle \textit{et al.}, 2003, 2006, 2007), the 30-year mean monthly precipitation used in the simulations was derived by intersecting the individual pixels with the Oregon State Climate Center prism data layer for precipitation. Aspect (energy) for each pixel was included with the vegetation data for each pixel.

The complete data set used in the hydrologic simulations consisted of the estimate of BA\textsubscript{max}, the dominant specie occupying the pixel, the total basal area present on the pixel, the monthly precipitation for the pixel, and the aspect of the pixel. Simulations were done by pixel and then aggregated to the watershed level. Hydrologic simulation was conducted on only the forested portion of each watershed. Non-forested areas were not considered in the simulations. Since the objective of the simulations focus on predicting the change in water yield, or water available for stream flow, and it is assumed that the change occurs as a result of insect mortality on the forested portion of the watershed, the non-forested portion was ignored. Simulations for each of the 8 watersheds of interest, as well the area in the North Platte River watershed above Seminole Reservoir not represented by any of the 8 watersheds, were completed using both the modal and the 95 percentile estimates of BA\textsubscript{max} for the conditions that existed in both 2002 and 2012. The change in stream flow was estimated as the difference between the 2002 and 2012 simulated seasonal stream flow. It should be noted that although scale varied considerably between watersheds, all simulations were done at the level of the pixel and aggregated and all pixels had the same data resolution.

4.2. Study Watershed Simulations

Simulated flow changes from the forested portions of the eight watersheds of interest and the area above Seminole reservoir were well correlated with the estimates of basal area change (Figure 14). The simulated flow changes depicted in Figure 14 are based on the modal estimate of BA\textsubscript{max}. The simulated changes in water yield based on the 95 percentile estimate of BA\textsubscript{max} were slightly lower (Figure 15) as might be expected. The WRENSS hydrologic model distributes the potential changes in the water balance, and water available for stream flow, over the range from BA\textsubscript{max} to BA\textsubscript{min}. The greater the estimate of BA\textsubscript{max}, the more gradual the slope of the change between any two estimates of basal area, thus a lower estimate of stream flow change with an increasing estimate of BA\textsubscript{max}. The R\textsuperscript{2}’s of the two simulated relationships (Figures 14 and 15) are similar as are the intercepts of the equations but the slopes of the lines differ. The regression equation for the modal distribution indicates a 2.2 mm increase in flow was simulated for each 1 percent reduction in basal area. This is 22 percent greater than the 1.81 mm increase estimated for the regression based on the 95 percentile simulation. Given that the 95 percentile value for BA\textsubscript{max} is quite high, the differences in the simulations indicate that having used the modal value for BA\textsubscript{max} seems reasonable. No further consideration will be given to the simulations using the 95-percentile estimate of BA\textsubscript{max}.
Perhaps more compelling, is the comparison of the simulated change in stream flow from the forested portion of the study watersheds with the relationship between basal area reduction and water yield increase observed to have occurred from the experimental watersheds as depicted earlier in Figures 3 and 5. In general, simulations of the effects of insect mortality for the study watersheds compare extremely well with observed responses from the experimental watershed studies as indicated by the comparison of the regression models for the two data sets (Figure 16). The simulations of flow change following forest disturbance resulting from insect mortality mimic what has been observed following timber harvest.

![Figure 14](image)

**Figure 14.** Simulated change in water yield based on the modal estimate of $\text{BA}_{\text{max}}$ and using the 30 year mean precipitation for the pixel.

Although the simulated responses compare well with the observed responses from experimental watersheds, the strongest validation of the merit of the simulations is in the stability and appropriateness of the relationship that exists between observed changes in water yield from the eight study watersheds and simulated changes in water yield for those watersheds (Figure 17). The simulated water yield changes ($Y$) plotted over observed water yield changes ($X$) based on the results of the ANCOVA analysis using the Glade Creek snow course (from Table 4) compare extremely well, with the exception of one outlier, North Brush Creek. The simulated change in flow represents the average increase in flow simulated to occur on the watershed once the full hydrologic effect of the mortality estimated from the vegetation layers has occurred. The observed change in flow represents the average value estimated to have occurred for the period 2002 to 2012 based on the results of the ANCOVA between the measured stream flow and the snowpack measured at the Glade Creek snow course. On average, peak infestation and subsequent tree mortality in the watersheds occurred in 2005 to 2007 so the flow changes that have occurred since that time, when averaged over the entire 2003 to 2012 period would underestimate the expected long term response. On average, the simulated increases in seasonal stream flow exceed observed increases by approximately 40 mm (1.6 in.).
Figure 15. Simulated change in water yield based on the 95-percentile estimate of BA\textsubscript{max} and using the 30 year mean precipitation for the pixel.

It should be noted that in the simulation process, no attempt was made to simulate the interim responses that occur as the vegetation cover goes from red to brown to grey followed by boles ultimately falling to the ground. Canopy interception will occur to varying degrees during that entire period reducing the net change in water available for stream flow. The potential effect will vary by aspect, basal area, and species composition. Our understanding about the impacts of canopy disturbance on interception would have allowed simulation of the change in this component of the water balance, but only at a resolution finer than our 10-year simulation increment. Pugh and Small (2011) found little difference in canopy interception between the green to red to brown phase. Canopy interception loss during the grey stage can be expected to be between what occurs in the green phase and that which occurs for aspen, given that aspen is deciduous and interception loss from the leafless canopy in the winter can be as high as 11 percent of the gross precipitation (interpreted from Figure 12). This would be an approximation, of course, but one with minimal error relative to measurement error. However, the potential effects on understory evaporation/sublimation are accounted for in the model as we assumed adequate roughness to retain snow on-site and the model otherwise accounts for some degree of increased sublimation at ground level.
Figure 16. Regression of the simulated water yield changes for the study watersheds and percent reduction in cover. Data was compared with the regression of the documented changes in water yield for the paired watershed studies and percent reduction in cover as presented earlier (Figures 3 and 5).

As part of the QA/QC aspect of stream flow assessment and simulation effort, every component in the analysis was verified as rigorously as possible. The vegetation data was the most difficult to verify but the validation that was possible supported the data layers used. First, only one watershed, Bear Creek, demonstrated a true decline in stream flow but that was supported by a 6 percent increase in basal area. Growing Stock Level (GSL) plot data from North Brush Creek collected as part of another study (personal communication with John Schmid, USFS retired) indicated that mortality ranged from 33- to 43-percent on the GSL study plots from 1999 thru 2008. The vegetation data layers used in this study indicate an average basal area reduction of 37 percent occurred on the North Brush Creek watershed. Although the GSL plot data in no way confirms the validity of the vegetation layers, it does not refute them. Also, all summary comparisons between the polygon data set used in the 2003 and 2006 reports compared well with the raster data set used in this study.
Figure 17. Simulated water yield change plotted over the observed water yield change estimated by the ANCOVA between water yield and the peak water equivalent measured at Glade Creek snow course.

The precipitation, snow course, and stream flow sites were also evaluated for any changes in procedures, location, and other factors that might cause change in either the measurement or the metric being measured. As noted earlier, the St. Vrain watershed was dropped from analysis because the stream gauge was moved during the 2002 to 2012 interval. The same was the case for the Saratoga, WY and Longmont, CO precipitation gauges as they also were moved or discontinued rendering the stability of the metric being measured suspect. Initially we were concerned about the estimated decline in flow observed for Bear Creek, but the estimated increase in vegetation density on the watershed mitigated that concern. The North Brush Creek and the Cache la Poudre River gauging sites have some issues that were addressed earlier.
4.3. Seminoe Reservoir Watershed Simulation

The contributing area above the Seminoe Reservoir consists of 22,042 km² (5,446,400 acres or 8510 square miles), of which 4,904 km² (1,211,803 acres or 1893 square miles) are forested and this represents 22 percent of the watershed area. As was done for the 8 study watersheds, the hydrologic simulation of water yield changes on the watershed addressed only that portion which is forested. Simulations were completed, by pixel, for 2002 and 2012 and then aggregated, by species, to the watershed level (Table 5) and presented earlier on Figures 14 and 15. As noted, the precipitation estimates used to drive the model are the 30-year mean monthly estimates obtained by intersecting the individual 240 X 240 m pixels with the latest Oregon State Climate Center prism precipitation layer. As such the simulated increase in water yield of 84.3 mm (3.32 in.) represents the average increase in seasonal water available for stream flow that might be expected to occur as a result of the changes in basal area that were estimated to have occurred between 2002 and 2012 on the forested portion of the Seminoe Reservoir watershed. This estimate of change in water yield represents what one might expect to occur when the full hydrologic impact of the estimated mortality (47 percent of the basal area on the forested portion) has been reached. The simulation may overestimate the actual stream flow increase observed to date because stream flow for the entire 10-year period was not fully affected by the mortality that did not occur until 2006 or 2007.

Table 5.
Simulated impact of vegetation changes on seasonal water yield entering the Seminoe Reservoir from the forested portion of the North Platte River watershed. Simulations used are based on hydrologic response to the 30 year mean monthly precipitation and, as such represents long-term average response.

<table>
<thead>
<tr>
<th>Seminoe Reservoir Watershed (Species Group)</th>
<th>Forested Area km² (acres)</th>
<th>Basal Area m²/ha (ft²/acre)</th>
<th>Mean Annual Precipitation mm (in.)</th>
<th>Simulated Flow Change mm (in.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Deciduous</td>
<td>761 (188,092)</td>
<td>2.7 (29.0)</td>
<td>2.66 (28.7)</td>
<td>587 (23.1)</td>
</tr>
<tr>
<td>Lodgepole pine</td>
<td>2,443 (603,570)</td>
<td>10.1 (108.6)</td>
<td>4.3 (46.2)</td>
<td>709 (27.9)</td>
</tr>
<tr>
<td>Ponderosa pine</td>
<td>80 (19,768)</td>
<td>4.7 (50.7)</td>
<td>2.3 (24.6)</td>
<td>538 (21.2)</td>
</tr>
<tr>
<td>Spruce–Fir</td>
<td>1,620 (400,373)</td>
<td>11.1 (119.4)</td>
<td>6.9 (74.8)</td>
<td>1,052 (41.4)</td>
</tr>
<tr>
<td>Weighted Watershed Mean</td>
<td>4,904 (1,211,803)</td>
<td>9.2 (98.8)</td>
<td>4.9 (52.6)</td>
<td>803 (31.6)</td>
</tr>
</tbody>
</table>
It can be noted (Table 5) that the specie groups are not equally represented aerially on the watershed and that the average precipitation varies considerably, by specie group. As one might expect, the least precipitation occurs in the Ponderosa pine/ Douglas-fir group and the greatest precipitation falls on the spruce-fir group. The simulated increase in flow was proportionally greatest for the spruce-fir group followed by lodgepole pine and then the Ponderosa pine/Douglas-fir group. There was little change in the Aspen component that the response cannot be evaluated. The greatest change in forest cover and the largest contribution to the total simulated increase in water yield occurred as a result of the 57.5 percent reduction in the basal area in lodgepole pine, which occupies the largest portion of the forested area. Troendle et al. (2006) speculated that as much as 90 percent of the saw timber and 50 percent of the pole sized lodgepole pine would be killed during the forecasted beetle infestation. Simulations of that scenario (Troendle et al., 2006) indicated the increase in seasonal water yield from the lodgepole pine type on the North Platte River would be 28,124 hectare-meters (228,000 acre feet). The simulated increase in yield from the lodgepole pine type of 107.7 mm (4.24 in.) as shown in Table 5 is equivalent to a 26,306 hectare-meters (213,000 acre feet) increase entering Seminoe Reservoir from the lodgepole pine type alone. A close comparison considering the vegetation data bases were from different sources. Spruce-fir mortality represented 37.4 percent of the basal area of that specie group (Table 5) although the unit area hydrologic response is greater than that for lodgepole pine. Increases in water yield following disturbance in spruce-fir can be expected to be proportionally higher than for other forest types. Spruce-fir has a greater leaf area index per unit of basal area than the other forest types causing it to evapotranspire more water so that the reduction in water use is greater following disturbance. In addition, spruce-fir generally receives greater precipitation than the other forest types and this also contributes to the greater response.

Overall, the estimated basal area reduction due to insect mortality has been quite significant in the forested portion of the Seminoe Reservoir watershed as is the simulated increase in water yield. The total simulated increase in seasonal water yield of 84.3 mm (3.32 in.) from the forested portion of the watershed represents 41,355 hectare-meters (335,265 acre feet) of water entering Seminoe Reservoir in an average year. In wet years the increase can be expected to be larger, in dry years it would be smaller, or perhaps non-existent. Given the annual variability of flow and the small percentage of forested area, on the watershed above the reservoir, the expected change in flow would be present but might be difficult to detect, even in wet years. Only 22 percent of the watershed is actually forested but the forest sustained 47 percent mortality, or decrease in basal area, from 2002 to 2012. Area weighted, mortality represents the equivalent of a 10 percent reduction in basal area at the watershed level. As Bosch and Hewlett (1982) noted, this would likely cause a change water yield but one below the level of detection. Most likely, the increases in stream flow would occur in May or June. Depending on the magnitude of the stream flow contribution from the 78 percent of the watershed that is not forested and was not influenced by insect mortality, the increase in water yield from the forested portion may or may not be detectable. We made no attempt to determine if the increase could be documented. Based on the performance of the WRENSS hydrologic model in simulating the stream flow changes from the 8 study watersheds, we feel confident the simulated increases will occur from the watershed above Seminoe Reservoir. A portion of the total increase has been documented at the North Platte, Encampment, and North Brush Creek watersheds are part of the watershed contributing to the Seminoe Reservoir.
4.4. Projecting Simulated Response into the Future

Projecting the effects of insect mortality on future water yield is a little more speculative in that it also requires that the future growth of vegetation be properly simulated. The Forest Vegetation Simulator (FVS) [Dixon 2002] is a forest growth and yield model designed to forecast forest stand development from standard forest inventory data. FVS grows individual forest stands into the future and considers factors such as the current stand conditions, regionally embedded growth and mortality relationships, and user-defined management options in the predictions. FVS involves post processing of multiple stand simulations to describe the average condition for a group of similar stands from which stratum-based yield tables are developed. Yield tables produced for multiple strata under multiple management options can then be used to obtain an optimal desired future condition. Our interest was in re-growing the stands as they existed in 2012 back to the pre-infestation condition that existed in 2002. FVS yield table development, for purposes of this study consisted of four developmental components: 1) forest inventory data were assigned to strata, 2) FVS was calibrated and adjusted for local conditions, 3) important tracking variables were identified and coded into FVS, and 4) natural growth runs were simulated that show forest growth patterns without active management over the simulation period, in this case 6 ten year increments. A fifth step would have been to introduce management options but that was not included in this effort. Keyser (2006) described all five steps in detail. Linear growth functions developed from yield tables, developed as part of the Shoshone National Forest plan revision process (Keyser, 2006), were considered appropriate and used to simulate the growth of the existing forest conditions present above Seminoe Reservoir in 2012 to return the watershed to the basal area levels that had existed in 2002. Vegetation growth was estimated for 6 ten year increments from the point of full mortality and the initial simulated increase in water yield was adjusted accordingly as vegetation cover, or basal area, increased over time. Robert Havis (USFS, Forest Management Service Center, Fort Collins, CO) assisted in developing the linear growth functions from the yield table data described by Keyser (2006). The slope of the function describing area weighted average basal area \(Y\) plotted over year \(X\) was used to increment the change in basal area over time. The linear functions developed by Havis indicated it would require 73 years for the basal area of lodgepole pine present in 2012 to grow back to the 2002 level. Sixty-seven years are required for the spruce-fir to recover and 35-years for Ponderosa pine/Douglas-fir. The re-growth functions for FVS were developed specifically for post-infestation conditions but no attempt was made in the growth projections to account for any additional mortality resulting from the continued spread of the infestation or for the occurrence of future management activities. Obviously, continued mortality would extend, or offset, the recovery and could increase hydrologic response. The simulated increases in water yield will decline gradually with time as the forest recovers (Table 6).

As noted earlier, it is assumed that the full hydrologic effect of insect mortality would not be reached for up to 15 years. This assumption was based on several factors. First, Bethlahmy (1974) concluded that it took 15 years for maximum hydrologic response to occur on the White River following a spruce beetle infestation. Second, any change in water yield during the green and red phase following insect mortality will be quite varied depending on aspect and precipitation. As noted, winter interception processes play a significant role in the hydrologic response and interception losses vary with aspect.
A greater percentage of the total potential changes in water yield will be realized earlier from higher energy sites, assuming the impact to the vegetation is significant enough to generate a change in water yield. Last, it was assumed that it would take an average of 15 years for the grey boles to fall. Once the stems are on the ground, the model addresses impacts of roughness, increases energy loading, and residual vegetation. The comparison between observed hydrologic responses from the study watersheds with the simulated hydrologic responses for the same watersheds would suggest the WRENSS hydrologic model, and the assumptions within, are appropriate and working well.

Table 6.
Decadal decline in the simulated increase in water yield in the Seminoe Reservoir

<table>
<thead>
<tr>
<th>Seminoe Reservoir Watershed (Species Group)</th>
<th>Simulated Water Yield Increase from Forested Area Following Mortality (mm)</th>
<th>Simulated Decline in the Water Yield Increase from Forested Area Over Time for Each 10-Year Period Following Mortality (mm)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Year 0 (in.)</td>
<td>Year 10 (in.)</td>
</tr>
<tr>
<td>Deciduous</td>
<td>1.0 (0.04)</td>
<td>0.0 (0.0)</td>
</tr>
<tr>
<td>Deciduous</td>
<td>0.0 (0.0)</td>
<td>0.0 (0.0)</td>
</tr>
<tr>
<td>Lodgepole pine</td>
<td>107.7 (4.24)</td>
<td>92.4 (3.64)</td>
</tr>
<tr>
<td>Lodgepole pine</td>
<td>92.4 (3.64)</td>
<td>77.5 (3.05)</td>
</tr>
<tr>
<td>Lodgepole pine</td>
<td>77.5 (3.05)</td>
<td>62.2 (2.45)</td>
</tr>
<tr>
<td>Lodgepole pine</td>
<td>62.2 (2.45)</td>
<td>47.2 (1.86)</td>
</tr>
<tr>
<td>Lodgepole pine</td>
<td>47.2 (1.86)</td>
<td>32.2 (1.27)</td>
</tr>
<tr>
<td>Lodgepole pine</td>
<td>32.2 (1.27)</td>
<td>17.3 (0.68)</td>
</tr>
<tr>
<td>Ponderosa pine</td>
<td>64.8 (2.55)</td>
<td>46.7 (1.84)</td>
</tr>
<tr>
<td>Ponderosa pine</td>
<td>46.7 (1.84)</td>
<td>28.4 (1.12)</td>
</tr>
<tr>
<td>Ponderosa pine</td>
<td>28.4 (1.12)</td>
<td>10.4 (0.41)</td>
</tr>
<tr>
<td>Ponderosa pine</td>
<td>10.4 (0.41)</td>
<td>0.0 (0.0)</td>
</tr>
<tr>
<td>Ponderosa pine</td>
<td>0.0 (0.0)</td>
<td>0.0 (0.0)</td>
</tr>
<tr>
<td>Ponderosa pine</td>
<td>0.0 (0.0)</td>
<td>0.0 (0.0)</td>
</tr>
<tr>
<td>Spruce-Fir</td>
<td>88.1 (3.47)</td>
<td>74.9 (2.95)</td>
</tr>
<tr>
<td>Spruce-Fir</td>
<td>74.9 (2.95)</td>
<td>61.7 (2.43)</td>
</tr>
<tr>
<td>Spruce-Fir</td>
<td>61.7 (2.43)</td>
<td>48.5 (1.91)</td>
</tr>
<tr>
<td>Spruce-Fir</td>
<td>48.5 (1.91)</td>
<td>35.3 (1.39)</td>
</tr>
<tr>
<td>Spruce-Fir</td>
<td>35.3 (1.39)</td>
<td>22.1 (0.87)</td>
</tr>
<tr>
<td>Spruce-Fir</td>
<td>22.1 (0.87)</td>
<td>8.9 (0.35)</td>
</tr>
<tr>
<td>Weighted Increase</td>
<td>84.3 (3.32)</td>
<td>71.9 (2.83)</td>
</tr>
<tr>
<td>Weighted Increase</td>
<td>71.9 (2.83)</td>
<td>59.4 (2.34)</td>
</tr>
<tr>
<td>Weighted Increase</td>
<td>59.4 (2.34)</td>
<td>47.24 (1.86)</td>
</tr>
<tr>
<td>Weighted Volume ha-m (acre-ft)</td>
<td>41,355 (335,265)</td>
<td>35,251 (285,783)</td>
</tr>
<tr>
<td>Weighted Volume ha-m (acre-ft)</td>
<td>35,251 (285,783)</td>
<td>29,148 (236,302)</td>
</tr>
<tr>
<td>Weighted Volume ha-m (acre-ft)</td>
<td>29,148 (236,302)</td>
<td>23,168 (187,829)</td>
</tr>
<tr>
<td>Weighted Volume ha-m (acre-ft)</td>
<td>23,168 (187,829)</td>
<td>17,314 (140,367)</td>
</tr>
<tr>
<td>Weighted Volume ha-m (acre-ft)</td>
<td>17,314 (140,367)</td>
<td>7,723 (62,610)</td>
</tr>
<tr>
<td>Weighted Volume ha-m (acre-ft)</td>
<td>7,723 (62,610)</td>
<td>5,730 (46,452)</td>
</tr>
</tbody>
</table>

The growth functions indicate that full hydrologic recovery, or return to the basal area levels present in 2002, would occur in about 70 years. This is slightly longer than the 60 years estimated for hydrologic recovery to have occurred on the Fool Creek watershed following timber harvest.
One potential problem in the simulation process resides with the fact that the hydrologic model is driven by basal area which is a surrogate for leaf area index. Although basal area in lodgepole pine type has been shown to be linearly correlated with leaf area index (Kaufman and Troendle 1981; Kaufman et al., 1982) we do not know if the relationship continues to be linear as residual stems and regeneration respond to the more enriched conditions that exist following partial mortality. Shifts in the leaf area index/basal area relationships from linear to nonlinear could result in a more rapid hydrologic recovery than basal area alone would predict. However, the time frame for the simulated recovery is quite reasonable given our understanding of the processes. In the 2003 report on the North Platte River basin, Troendle et al. used stand inventory data from R2VEG and simulated that it took similar lengths of time for stands of the same species groups to grow from the sapling and small pole size classes into the large pole and saw timber classes, or those stands at greatest risk to insect mortality.

5. Discussion

This study has primarily focused on the effects of forest disturbance on water yield measured at the stream gauge at both the small watershed and landscape scale. Minimal linkages have been presented to the myriad of plot and process studies done in conjunction with the watershed studies reported here or those process, plot, and watershed studies currently being conducted and documented by others. This was done purposely because as Hewlett et al. (1967), Reynolds and Leyton (1967), and Pereira (1962) concluded decades ago; watershed response must always remain the ultimate proof of hydrological conclusions drawn using other techniques.

The current insect infestation has had, and will continue to have, landscape level impacts on forest vegetation, primarily in the spruce-fir and lodgepole pine types. Defining what can be demonstrated to have occurred in measured stream flow at the watershed or landscape level provides a context in which the plot and process studies, both current and historic, can be assessed and used to interpret and further extrapolate the observed watershed responses. The general pattern of stream flow response to forest disturbance, primarily timber harvest, demonstrated to have occurred worldwide (Bosch and Hewlett 1982) encompasses the documented hydrologic responses following timber harvest observed to have occurred following paired watershed experiments in the Rocky Mountain region. Although there is a paucity of studies worldwide documenting the impacts of either insect or fire effects on water yield, the two documented responses from the Rocky Mountain region compare well with the paired watershed responses and provide a context or framework for further study into the effects of insect driven tree mortality on water yield. The stream flow response following basal area reductions, presumably the result of insect mortality, on 8 study watersheds in the North and South Platte River watersheds also appear to follow a similar trajectory to that of timber harvest; a change in forest cover results in a change in water yield. The documented hydrologic responses on the 8 insect infested watersheds are not overly dramatic and not all 8 watersheds demonstrated a significant increase in water yield, but the regression of increasing water yield over reduction in cover for the 8 watersheds was significant and similar in slope to the regression for the paired watershed studies.
Three of the 8 study watersheds demonstrated a significant increase in water yield that is difficult to refute. One of these 3 watersheds is regulated, but 2 of the 3 represent the only suitable and unregulated watersheds available for analysis in the Platte River Basin. Of the 5 remaining study watersheds, all but Bear Creek are heavily regulated. We were unable to determine, as part of this study, whether or not the historical operational hydrology practices that appeared consistent in the watershed selection process have been maintained in recent years on those 5 watersheds. It has been well documented that May is the only month in which detectable increases in stream flow can be expected to occur in the Rocky Mountain region. If opportunistic diversions to satisfy junior water rights have increased during May in response to any increased flow, or what may have been interpreted as early snow melt, that flow change would not pass the stream gauge and would go undetected. Also, annual precipitation has generally been low in recent years and Eq. 2 would suggest that at least 460 mm (18.1 in.) of precipitation would have to occur before a measureable increase in flow might be expected in response to forest disturbance. Stednick and Jenson (2007) suggested that the minimal precipitation requirement is slightly higher at 508 mm (20 in.). Not all of the 8 study watersheds have received that much precipitation in each of the recent years, further confounding the opportunity for a change in water yield to occur and be documented.

Based on all 95 paired watershed studies conducted in the United States, Stednick (1996) determined that the increase in water yield following timber harvest averaged 2.5 mm (0.10 in.) for each 1 percent reduction in cover. Stream flow increases following timber harvest in the Rocky Mountain region, depicted on Figure 3, represents a 2.3 mm (0.09 in.) increase in water yield for each 1 percent reduction in cover. The slope of the regression model fit to the documented increases in water yield for the 8 study watersheds (Figure 9) reflects a 2.2 mm (0.09 in.) increase in stream flow following a 1 percent reduction in cover. The documented increases in stream flow estimated to have occurred on the 8 study watersheds as the result of insect mortality appear defensible, are consistent with other documented responses following timber harvest and other forest disturbances, and most likely reflect conservative estimates of the potential increase in stream flow that will occur over time.

The net effect of any form of forest disturbance, whether timber harvest, fire, wind throw, or insect mortality has on the water balance is best characterized by what is measured at the stream gauge. Plot and process studies characterize site specific components of the water balance but it is only at the stream gauge that the variability in site specific responses are integrated across the watershed and the net change in water yield documented. Michelson et al. (2013) have summarized much of the recent research in the Rocky Mountain region on the effects of insect mortality on the components of hydrologic response. For the most part, the body of work consists of plot studies that would indicate insect mortality will potentially decrease net ET at the watershed level and potentially result in an increase water yield at the watershed level, with little quantification presented. Most recent studies have defined differences that exist between plot pairs, not necessarily the change that has occurred between the plots. Insect mortality has been so widespread and rapid, there has been little opportunity to establish adequate controls to document and characterize pre-infestation conditions to the point where post infestation change can be documented.
For example, the effect that timber harvest has on water yield is strongly dependent on the effect of canopy cover reduction on snow pack accumulation (Figure 12). Pugh and Small (2011), and Boon (2007, 2009) among others have documented that differences in snow pack accumulation exist between live and dead stands following insect mortality. Pugh and Small (2011) compared snowpack accumulation under 8 paired living and dead stands. They were unable to detect change in snow pack accumulation as the stand transitioned from the green to the red phase in lodgepole pine. They did observe an 11 to 21 percent difference in accumulation under the grey phase stands when compared to the live stands. The variability in plot characteristics they presented, however, would suggest significant differences in snow pack accumulation would have existed between plots prior to infestation so their snapshot observation following infestation represents detection of a difference but not necessarily documentation of the actual change that may have occurred. In general, the fact that most current studies assessing snow pack accumulation are relegated to looking at post-beetle infestation, differences can be documented but assessing change is more complex.

Insect mortality results in a partial reduction in basal area while leaving the un-infested trees intact. Following infestation, the residual stand has less competition for the available resources on site. Rhoades et al. (2013) and others have proposed the concept of compensatory use of resources by the residual vegetation. The implication is that following insect mortality, the residual vegetation responds quickly to utilize the water and nutrient resources at a rate greater than would occur following a commensurate level of timber harvest, or thinning; thus minimizing or even negating the potential hydrologic response or increase in water yield. However, increases in resource use by residual vegetation are also well documented following mechanical thinning. Hypotheses related to the effect of compensatory use of resources following insect mortality on subsequent water yield would need to be evaluated, relative to those following mechanical forest thinning before inferences can be drawn that the allocation processes and the ultimate effect on water yield would differ following mechanical thinning and insect mortality. For example, Troendle (1987a) reported on a replicated thinning study conducted in 60 to 70 year old lodgepole pine in Colorado. The study design consisted of 5 replicated blocks with 4 thinning levels imposed per block. One block was treated in each of 5 successive years providing an adequate mix of pre- and post-treatment comparisons. Soil moisture content in the 1.5 m soil profile was measured bi-weekly on all plots during the growing season and snow water equivalent was measured on or about April 1 each year on each plot. Data was collected over a 10 year period. It was documented that on average the soil water depletion per unit of leaf area index in the residual stands increased exponentially from 0.25 mm (0.01 in.) to 0.51 mm (0.02 in.) per day as basal area was reduced from 27.5 m² per ha. (120 ft.² per acre) to 9.2 m² per ha. (40 ft.² per acre). In addition, analysis also indicated soil water depletion during the growing season was positively correlated with basal area only in average and wet years while basal area was not significantly correlated with soil water depletion in below average precipitation years; confirming numerous observations that in dry years all available water is used by the residual vegetation regardless of cover, even when the site has been clear-cut. The measured April 1 water equivalent on the plots increased linearly with decreasing basal area at a rate similar to the east west relationship presented in Figure 12. Complete ablation of the snow pack occurred 7 to 14 days earlier under the least cover.
A far more comprehensive study of the components of the water balance, replicated in mature lodgepole pine with a spruce-fir understory, was reported by Wilm and Dunford (1948). They estimated that an average net decrease in ET of 81 mm (3.2 in.) occurred on the clear-cut plots relative to the uncut controls. The average reduction in net ET estimated for plots in which 50 percent of the forest cover had been removed was 28 mm (1.1 in.) or 34 percent of that observed for the clear-cuts. ET reduction averaged 51 mm (2.0 in.) when 67 percent of the cover was removed and 53 mm (2.1 in.) when 83 percent of the cover was removed. The reduction in ET that occurred as the forest cover was decreased was nonlinear. The residual vegetation consisted of mature (200 + year old) overstory and a mixed species understory, which responded to the increase in available water.

The works of Pugh and Small (2011), and Rhoades et al. (2013) are representative of the studies documenting differences perceived to exist between the hydrologic response following timber harvest and that which occurs following insect mortality. Obviously, when trees go through a process of dying followed by the canopy transitioning from green to red to brown to grey with the boles ultimately falling to the forest floor over a period of a few years, there will be short term differences in response in comparison to what happens if the tree is severed and the bole immediately loaded on a truck. These studies substantiate the commonality that exists between timber harvest and insect mortality, and were considered in the simulations. Mikkelson et al. (2013) summarized recent research on the effects of the bark beetle infestation on hydrologic response. There are findings, mostly from uncontrolled plot studies, that will support or refute virtually any conclusion regarding the effects of insect morphology on hydrologic response. However, in summarizing their review of the science Mikkelson et al. (2013) drew 4 conclusions about hydrologic response following insect mortality:

- While the magnitude of the shift is variable, snow depth will increase in infected catchments.
- Changes in ET are more variable and may be offset by competing components such as decreased transpiration and increased ground evaporation.
- Soil moisture increases are probable, although seasonal fluctuations may be influential in determining overall increases.
- Stand-scale water yield will increase; however, confounding factors such as percent infested, evaporation rates, and climate make predicting changes in water yield difficult on a watershed or regional scale.

The first three conclusions drawn by Mikkelson et al. (2013) are equally true, and have been well documented, for the effects of timber harvest on water yield (Hibbert 1967). In summarizing the effects of forest treatments on water yield, Hibbert (1967) also concluded that response was variable and difficult to predict. The fourth conclusion drawn by Mikkelson et al. (2013) is quite similar to that of Hibbert (1967) but the ability to predict response has improved significantly since 1967. We need to focus more on the similarities that exist between timber harvest and insect mortality and in the process, the knowledge gap and research needs will become more obvious.
The documented changes in water yield that have been estimated to have occurred from the 8 study watersheds in response to insect mortality seem reasonable and defensible from an experimental standpoint, and consistent with the body of ongoing work, whether associated with timber harvest or insect mortality. It would also appear that the hydrologic simulation of the observed changes is also reasonable and the inference can be drawn that the WRENSS hydrologic model is capable of simulating the potential effect of insect mortality on stream flow from ungauged watersheds. The WRENSS hydrologic model simulations did not include conditions specific to the transition phases of the impacted stands. Any error would be associated with estimating the interim winter interception losses and would be greatest on north slopes and least on south slopes. Although transpiration will cease immediately following insect attack (Hubbard et al., 2013), interception will not significantly decrease for 2 or 3 years following mortality causing only a partial hydrologic response. It is reasonable to assume that the simulated increase in flow of 84.3 mm (3.32 inches) from the 4,904 km² (1,211,800 mi.²) forested portion of the Seminoe Reservoir watershed reasonably reflects what might be expected to occur. The unknown is the length of time that will be required to achieve that increase. We have assumed 15 years from infestation, which is consistent with the observation by Bethlahmy (1974) for the White River, but it could be achieved sooner.

Simulating the time frame during which hydrologic recovery will occur as the stands re-grow following infestation is the most speculative aspect of the study. Hydrologic recovery was estimated using growth projections based on the FVS vegetation model. As implemented for this study, the growth functions used in FVS were developed to be responsive to post infestation vegetation and site conditions. FVS projects basal area change (growth) over time. Hydrologic recovery has been expressed as the length of time required to return the existing forest condition to the basal area levels that existed prior to infestation. Since it is assumed that the leaf area/basal area relationship is the same for the recovering stand as it was for the original stand, a linear recovery is simulated. The only data available indicates that the leaf area/basal area relationship is linear, so hydrologic recovery following the recent mortality can be expected to occur in 60 to 70 years.

6. Summary

The effect of reductions in basal area, resulting from insect mortality, have documented to increase seasonal water yield from watersheds ranging from 160 km² (62 mi.²) to 3,700 km² (1429 mi.²) in the Platte River basin. Simulation of water yield change, using the WRENSS hydrologic model compared well with the observed responses over the entire range, with the exception of two outliers and it would appear the discrepancy lies with the field data more so than the simulations. It is reasonable to assume that the simulated increase in flow of 84.3 mm (3.32 in.) from the 4,904 km² (1,211,800 mi.²) forested portion of the Seminoe Reservoir watershed reasonably reflects what might be expected to occur as a result of the reduction in basal area estimated to have occurred to date. In all likelihood the actual increases in water yield, from past as well as future insect related mortality, will be greater than simulated. All simulations were run with adequate roughness in place to retain the snow and avoid scour thereby maximizing the estimate of water yield increase.
It should be noted that the average change in stream flow that might be expected to occur, as presented in Table 6, are based on simulations using estimates of the average 30 year mean monthly precipitation. The actual response that can be expected to occur in any given year will vary and be dependent on the precipitation that occurs in that particular year. In average and wetter years, the increase in water yield can be expected to equal or exceed the simulated estimates. In dry years, an increase in water yield may not occur.

WRENSS does not allow the distribution of the change in water yield over the runoff season. However, as noted earlier, May is the only month of the year during which water yield was consistently and significantly increased following timber harvest and it should be safe to assume the timing of the flow change will be similar following insect driven mortality. There are a few instances when water yield was significantly increased during June but water yield during all other months was not detectably impacted. On average, approximately 50 percent of the annual increase in water yield should occur in May, about 25 percent occurs in June and the balance during the rest of the runoff period.
7. References


Swanson, R.H. 2004. The complete WRENSS hydrologic model. Proceedings of the 72nd Western Snow Conference. Richmond, B.C.


