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Key Points:
- Recovery times of stream runoff and nitrate concentration after forest disturbance were defined and compared using data from 20 sites
- A linear correlation between runoff and nitrate recovery times was significant for catchments where impacts of runoff were observed
- Stream runoff recovery times were longer than nitrate recovery times after extreme forest disturbance

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Stream Runoff and Nitrate Recovery Times After Forest Disturbance in the USA and Japan

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Abstract To understand mechanisms of long-term hydrological and biogeochemical recovery after forest disturbance, it is important to evaluate recovery times (i.e., time scales associated with the return to baseline or predisturbance conditions) of stream runoff and nitrate concentration. Previous studies have focused on either the response of runoff or nitrate concentration, and some have specifically addressed recovery times following disturbance. However, controlling factors have not yet been elucidated. Knowing these relationships will advance our understanding of each recovery process. The objectives of this study were to explore the relationship between runoff and nitrate recovery times and identify potential factors controlling each. We acquired long-term runoff and stream water nitrate concentration data from 20 sites in the USA and Japan. We then examined the relationship between runoff and nitrate recovery times at these multiple sites and use these relationships to discuss the ecosystem dynamics following forest disturbance. Nitrate response was detected at all study sites, while runoff responses were detected at all sites with disturbance intensities greater than 75% of the catchment area. The runoff recovery time was significantly correlated with the nitrate recovery time for catchments that had a runoff response. For these catchments, hydrological recovery times were slower than nitrate recovery times. The relationship between these two recovery times suggests that forest regeneration was a common control on both recovery times. However, the faster recovery time for nitrate suggests that nitrogen was less available or less mobile in these catchments than water.

1. Introduction

Societies rely on a consistent flow of ecosystem services from forests; thus, we need accurate predictions of how those services will be affected by disturbance events. Ecosystem services are multidimensional, and thus a major challenge is understanding the recovery—or resilience—of multiple ecosystem functions after disturbance. Recovery times have been proposed as a way to quantify resilience (Walker et al., 2004), and thus, we have an opportunity in forest science to start assessing multiple recovery times after disturbances—particularly the hydrological and biogeochemical responses and recovery times to predisturbance levels.

Many previous studies have analyzed the hydrological and biogeochemical responses of catchments to disturbance, and a number of summaries have been published (Binkley & Brown, 1993; Bosch & Hewlett, 1982; Creed et al., 2014; Hornbeck et al., 1993; Jones et al., 2012; Stednick, 1996; Vitousek et al., 1979). The recovery of runoff and biogeochemistry at the catchment scale has been assessed with paired catchments for many decades (Bormann et al., 1969; Brown et al., 2005; Sebestyen et al., 2011; Swank & Vose, 1997; Vose et al., 2014). Around the world, natural disturbances (e.g., fire, diseases, and insects) have been documented in reference basins in areas with different climate, vegetation, and soil (Amatya et al., 2016; Argerich et al., 2013; Jones et al., 2012). Previous reviews have addressed the annual water yield increases after reduction of forest cover (Bosch & Hewlett, 1982; Brown et al., 2005; Stednick, 1996; Zhang et al., 2017). The maximum water yield within 5 years of deforestation (Bosch & Hewlett,

1982) and the initial runoff responses are greatest in high rainfall areas but are shorter where revegetation is rapid (Stednick, 1996).

Nitrogen is a vital element for forest productivity and thus a commonly reported solute in previous studies of forest disturbance. Excess nitrogen leaching, mainly as nitrate, usually occurs following forest disturbance (Likens et al., 1970), and increased nitrate leaching has been documented in response to cutting (Bormann & Likens, 1979; Cummins & Farrell, 2003; Martin et al., 2000; Mupepele & Dormann, 2017; Oda et al., 2011; Reynolds et al., 1995), fire (Knoepp & Swank, 1993; Meixner et al., 2003; Riggan et al., 1994), and insect defoliation and disease (Eshleman et al., 1998; Swank et al., 1981; Tokuchi et al., 2004). The initial nitrate concentration increase has been correlated to disturbance intensity (Wang et al., 2006).

Forest disturbance influences runoff and stream water nitrate concentration, but recovery times of runoff and nitrate concentration after disturbance have been less studied in a synthesis study of many sites. Hydrologic recovery time has been defined as the time required for annual runoff yield to return to pretreatment level (Hibbert & Gottfried, 1987; Stednick, 1996). Some studies have shown an increase in runoff to occur upon harvesting, with a rapid return to the predisturbance state (Brown et al., 2005; Hornbeck et al., 1993), although other studies have shown recovery to last for decades (Troendle & King, 1985). Much of the prior work on runoff recovery time has addressed how recovery relates to basal area removed and the regrowth of vegetation (Adams et al., 2014; Bosch & Hewlett, 1982; Swank & Douglass, 1974; Verry, 1976; Vitousek & Reiners, 1975).

Similarly, nitrate concentration usually increases after forest disturbance (Aber et al., 1989; Eshleman et al., 1998; Likens et al., 1970) and declines to predisturbance levels rapidly as vegetation regrows (Bormann & Likens, 1979; Martin et al., 2000; Vitousek, 1977), although some catchments exhibited little response in nitrate concentration (Rhoades et al., 2013; Sebestyen & Verry, 2011). Some studies have shown that stream water nitrate concentration recovery time is related to hydrology (Riscassi & Scanlon, 2009) and disturbance type (Lovett et al., 2002; Riscassi & Scanlon, 2009).

In previous studies, it was found that increased stream runoff and nitrate concentration after forest disturbance recover with vegetation regrowth. If the controlling factors of runoff and nitrate concentration recovery times are common, it is expected that recovery time of runoff and nitrate leaching following forest disturbance should be similar. While many studies have addressed either runoff or nitrate concentration recovery after disturbance, much less is known about how these two factors are related. To address this knowledge gap concerning ecosystem recovery following disturbance, it is necessary to gather and compare stream runoff and nitrate concentration responses and recovery times after forest disturbance in many catchments with different climates, vegetation covers, geologies, disturbance types, and forest manipulations. How these time scales of recovery correspond, and what factors influence this correspondence, is the focus of the current study.

In this study, we analyzed data from 20 Japanese and U.S. catchments with long-term measurements of stream runoff and nitrate concentration to assess time to recovery after forest disturbance. We examined paired catchment approach and before-after experimental designs to characterize runoff and nitrate responses and recovery times to evaluate the efficacy of some approaches relative to paired-catchment studies.

2. Methods

2.1. Sites and Data

We included temperate forested catchments in Japan and the United States, with precipitation ranging from 700 to 1800 mm per year and mean annual temperatures ranging from 3.4 to 14 °C. The forests range from mostly deciduous to mostly coniferous with some mixed cover types, with vegetation under various levels of management (Table 1 and Figure 1). We classified forest disturbance type, as clearcutting, harvesting and some additional management (Cut+), clearcutting and 100% cutting by the accumulation of multiple partial cuts (Cut), partial cutting (Partial cut), and tree mortality and defoliation due to insect and disease (Defoliation). Additional management includes understory vegetation removal and herbicide application. Since this intensity of disturbance and the effects may be differ from overstory-only clearcutting, we separated Cut+ from Cut. The catchments included in the analysis are described elsewhere (Adams et al., 2012;

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Table 1
Summary of Characteristics of 20 Forested Catchments

Site	Location	Area (ha)	Mean annual temperature (°C)	Mean annual precipitation (mm/year)	Vegetation	Disturbance type (classification)	Disturbance area (%)	Disturbance year	Runoff data length (interval)	Nitrate data length (interval)	Water year	Paired (Y/N)
USA												
Hubbard Brook W2	43°56′N, 71°45′W	19	6	1,400	Northern Hardwood	Clear felling and herbicide (Cut+)	100	1965	1955–2012 (daily)	1963–2012 (weekly)	June–May	Y
Hubbard Brook W4		43	6	1,400	Northern Hardwood	Sequential strip cuts (Cut)	100	1965	1955–2012 (daily)	1963–2012 (weekly)	June–May	Y
Hubbard Brook W5		21.9	6	1,400	Northern Hardwood	Whole-tree harvest (Cut)	100	1983	1955–2012 (daily)	1970–2012 (weekly)	June–May	Y
Marcell S4	47°32′N, 93°28′W	34	3.4	780	Upland aspen/birch; black spruce on peatlands	Partial harvest (Partial cut)	76	1970–1972	1961–2010 (daily)	1971–2010 (biweekly)	October–September	Y
Marcell S6		8.9	3.4	780	Upland aspen/birch converted to spruce/red pine	Partial harvest (Partial cut)	78	1980–1983	1976–2010 (daily)	1977–2010 (biweekly)	October–September	Y
Few now WS1	39°02′N, 79°40′W	29.9	9.2	1,458	Mixed hardwoods	Clearcut (Cut)	100	1958	1951–2009 (daily)	1971–2009 (weekly)	January–December	Y
Fewnow WS2		15.4	9.2	1,458	Mixed hardwoods	Partial harvests (Partial cut)	75	1988	1951–2009 (daily)	1983–2009 (weekly)	January–December	Y
Fewnow WS3		34.4	9.2	1,458	Mixed hardwoods	Partial cut Clearcut (Cut)	100	1958–1963, 1970	1951–2009 (daily)	1971–2009 (weekly)	January–December	Y
Fewnow WS5		36.4	9.2	1,458	Mixed hardwoods	Partial cuts (Partial cut)	86	1988	1951–2009 (daily)	1983–2009 (weekly)	January–December	Y
Fewnow WS6		22.3	9.2	1,458	Norway spruce	Clearcut and herbicided (Cut+)	100	1964–1969	1951–2009 (daily)	1971–2009 (weekly)	January–December	Y
Fewnow WS7		24.3	9.2	1,458	Mixed hardwoods	Clearcut and herbicided (Cut+)	100	1963–1969	1951–2009 (daily)	1971–2009 (weekly)	January–December	Y
North Fork Dry Run	39°19′N, 77°43′ W	230	14	1,357	Mixed deciduous	Gypsy moth (Defoliation)	60	1988–1992	1988–2010 (daily)	1987–2010 (weekly)	October–September	N
White Oak Run		510	14	1,357	Mixed deciduous	Gypsy moth (Defoliation)	93	1988–1992	1979–2010 (daily)	1979–2010 (weekly)	October–September	N
Paine Run		1240	14	1,357	Mixed deciduous	Gypsy moth (Defoliation)	35	1988–1992	1992–2010 (daily)	1992–2010 (weekly)	October–September	N
Piney River		1260	14	1,357	Mixed deciduous	Gypsy moth (Defoliation)	59	1988–1992	1992–2010 (daily)	1992–2010 (weekly)	October–September	N
Staunton River		1050	14	1,357	Mixed deciduous	Gypsy moth (Defoliation)	85	1988–1992	1992–2010 (daily)	1992–2010 (weekly)	October–September	N
Japan												
Fukuroyamasawa	35°12′N, 140°06′E	1	14	2,230	Conifer (Japanese cedar and cypress)	Clear cut (Cut)	100	1999	1994–2014 (hourly)	1998–2014 (weekly-monthly)	January–December	Y

Table 1 *(continued)*

Site	Location	Area (ha)	Mean annual temperature (°C)	Mean annual precipitation (mm/year)	Vegetation	Disturbance type (classification)	Disturbance area (%)	Disturbance year	Runoff data length (interval)	Nitrate data length (interval)	Water year	Paired (Y/N)
Kiryu M	34°58′N, 136°35′E	0.68	13.5	1,631	Conifer (Japanese cypress)	Pine Wilt Disease (Defoliation)	25	1994	1990–2012 (daily)	1990–2012 (biweekly)	January–December	N
Oyasan	36°33′N, 139°21′E	1.8	12	1,750	Conifer (Japanese cedar and cypress)	Partial cutting (Partial cut)	18	2000	1978–2012 (daily)	1978–2012 (weekly)	November–October	N
Teshio	45°03′N, 142°06′E	8	5.6	1,170	Mixed conier and deciduous	Clearcut and understory vegetation cut 3(Cut+)	100	2003	2000–2010 (monthly)	2002–2010 (biweekly)	October–September	N

Fukuzawa et al., 2006; Likens & Buso, 2012; Oda et al., 2009; Ohte et al., 2003; Riscassi & Scanlon, 2009; Sebestyen et al., 2011; Urakawa et al., 2012).

In general, precipitation was measured using tipping bucket rain gages, recording rain gages (e.g., Universal Recording Precipitation Gages; Brakensiek et al., 1979), and standard rain gages (US Weather Bureau, 1913) or was estimated using PRISM data set (Daly et al., 2002). Annual runoff (RO) as specific discharge was determined by summing daily stream discharge values and dividing by the catchment area. The water year used for summing annual runoff was varied by site (Table 1) to minimize interannual changes in storage.

Stream water nitrate concentration at each site was typically monitored with grab samples collected on a weekly to monthly basis (Table 1). Nitrate at each site was measured by ion chromatography or automated colorimetry on flow injection analyzers. Annual volume-weighted concentrations were calculated on a water year basis by linearly interpolating stream water concentration samples to estimate daily values.

2.2. Data Analysis

2.2.1. Detection of Hydrological Change Following Forest Disturbance

To evaluate the hydrological change following disturbance, we used RO and annual precipitation (*P*). The interannual variability of *P* and RO makes hydrologic change detection difficult. Thus, many forested catchment studies use the paired catchment approach to optimize the detection of RO change due to experimental treatment (Bosch & Hewlett, 1982). We used four methods to evaluate hydrological change and recovery time. One is the paired catchment approach. Our study includes some catchments that do not have reference catchments; thus, we also used three alternative methods that characterize the hydrologic variability prior to disturbance and compared them to postdisturbance variability to detect hydrologic change.

We implemented the paired catchment approach for sites that had reference catchments. Regressions of RO between experimental and reference catchments were used. First, we calculated the difference during a predisturbance period, to provide a baseline of a difference between two catchments without disturbance. We calculated the mean + 2 SD (standard deviation) of the difference in RO between the catchments during the predisturbance period and compared those values to the postdisturbance values. Since the annual runoff difference has year-to-year fluctuation even when there is no disturbance, this influence must be taken into account when evaluating the effect of disturbance. Therefore, in this study, the mean + 2 SD of difference of the predisturbance period was adopted as the most restrictive criterion of detection. If any differences were larger than the predisturbance mean + 2 SD, we considered an RO change to have taken place. The year-of-peak RO change, when RO difference between experimental sites and reference is its maximum, was determined, and the RO peak time, which is the time between the year-of-peak disturbance and the year-of-peak RO change, was calculated. The RO recovery time was considered to be the time between the peak forest disturbance and the first year when the difference was the mean of predisturbance values +5% of peak RO change (95% recovered). The analyses for catchments at Fernow, Fukuroyamasawa, Hubbard Brook, and Marcell were paired-catchment assessments (Method 1).

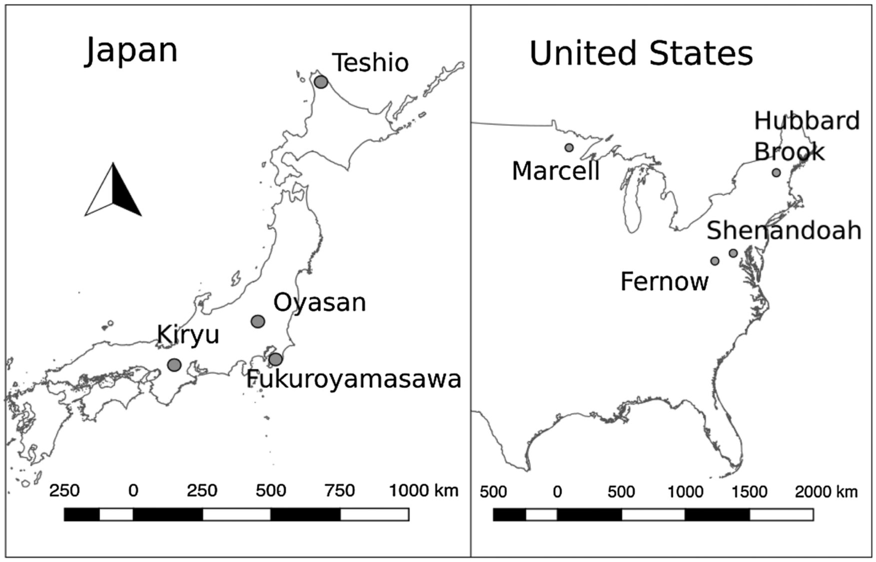

Figure 1. Map of the catchments used in this study from Japan and the United States.

When a reference catchment was unavailable, we used the linear relationship between annual *P* and RO during the predisturbance period to estimate a posttreatment RO deviation (RO = $aP + b$) (Method 2, a and b are fitting parameters) and annual runoff ratios (RO/P) (Method 3) and hydrologic loss (P-RO, Method 4) to assess the RO response and recovery. Method 2 required at least 4-year predisturbance *P* and RO data. The linear relationship between *P* and RO was used to estimate the expected runoff in a precipitation scenario during the postdisturbance period if no disturbance had taken place. If the postdisturbance deviation from the expected RO was greater than the mean + 2 SD of the residuals of that relationship, a forest disturbance response was considered to have occurred. The timing of the peak RO response was calculated as the difference between the year-of-peak forest disturbance and the peak RO deviation. The hydrologic recovery time was defined by the time between peak disturbance and the first occurrence of 95% recovery. For Method 3 and Method 4, we used all predisturbance data and the predisturbance mean + 2 SD and mean values were calculated. The mean + 2 SD value was compared to the posttreatment data to determine whether an RO response occurred. If an RO response was detected, the peak RO response was calculated by using the mean annual precipitation to estimate the RO deviation. RO recovery time was defined by time from peak disturbance and the first occurrence of 95% recovery to predisturbance mean value.

Since the detection of RO change included eight catchments with no paired reference catchments, we decided to test the robustness of Methods 2–4, which were applied to the unpaired catchments. To do this, we applied Methods 2–4 to the population of catchments that were paired and compared the results with those yielded by Method 1, which is assumed to be the most reliable method (Hewlett, 1971; Wilm, 1944; Zhao et al., 2010).

2.2.2. Detection of Stream Water Nitrate Change Following Forest Disturbance

To evaluate the change of stream water nitrate concentration following disturbance, we used volume-weighted annual nitrate concentration and absolute amounts of annual nitrate export. Two methods (Method N1 and Method N2) were used to assess the nitrate response magnitude and recovery time—neither was reliant on a paired catchment approach. We also compared the recovery times with concentration and absolute amount of nitrate export. The ratio of SD during the predisturbance period to the peak response averaged 17% for RO, on the other hand, the same ratio averaged 5% for nitrate, and the interannual variation of nitrate concentrations was generally less than that of RO. Nitrate response was much clearer than RO response; thus, less predisturbance data were needed to detect a signal. In method N1, we used predisturbance data to calculate the mean and the mean + 2 SD value of nitrate concentration prior to disturbance. If the postdisturbance nitrate peak concentration was larger than the predisturbance mean + 2 SD value, we considered a nitrate change to have taken place. The year of peak nitrate change, in which the deviation of nitrate concentration between experimental and reference sites was maximum, was determined, and the nitrate peak time, which is the time between the year-of-peak disturbance and the year-of-peak nitrate change, was calculated. Then the magnitude of response (difference between mean predisturbance nitrate concentration and peak nitrate concentration after disturbance) was calculated. The nitrate recovery time was considered the time between the peak forest disturbance and the first year when nitrate concentration recovered to the mean predisturbance value +5% of peak response (95% recovered).

If no predisturbance data were available, we used the stable nitrate concentration during postdisturbance years to estimate the predisturbance mean and maximum values (Method N2). The data showed that SD before disturbance was 5% or less of peak response. The stable nitrate concentration after disturbance was assumed to be the 5-year mean concentration when 5-year SD was smaller than 5% of the difference between peak concentration and 5-year mean concentration (5-year SD/[(peak concentration − 5-year mean] < 0.05). After reaching the stable concentration, the maximum value of nitrate concentration in the prerecovery period was determined. Sites with predisturbance data showed that the predisturbance and postdisturbance nitrate concentrations were similar enough to use the postdisturbance data to estimate predisturbance values.

The pretreatment period was as short as 1 year at Fukuroyamasawa, Marcell S4, North Fork Dry Run, and Teshio. The sites used for Method N2 were Fernow WS1, WS3, WS6, WS7, Fukuroyamasawa, Teshio, Marcell S4, North Fork Dry Run, Piney River, Staunton River, and Paine Run. For Marcell S4, Fernow WS 1 and WS 3, nitrate data were only measured after disturbance, and due to the influence of urea application or fertilization after cutting, it was impossible to estimate nitrate recovery time.

2.2.3. Comparison of Disturbance Response to Site Characteristics

We compared the hydrologic and nitrate response and recovery times to site disturbance types, intensity, and climatic characteristics to gain insight into the sensitivity of responses and recovery times to external forcing. The disturbance types included Cut, Cut+, Partial cut, and Defoliation. The climatic characteristics included mean annual temperature and precipitation and were used to evaluate whether the responses to disturbance might be sensitive to climate. Two linear mixed effects models were built to assess the effects of site disturbance intensity and climatic characteristics on the magnitude of the peak response and the recovery time using the nlme package in R (Pinheiro et al., 2017, R package version 3). The peak impact mixed effects model used response type (runoff or nitrate), disturbance type, mean annual temperature, and mean annual precipitation as fixed effects and used location as a random effect due to multiple experiments from the same location (e.g., Hubbard Brook). Due to the different units of the nitrate and runoff peak impacts, the peak nitrate and runoff from each study were made comparable by converting to an effect size before being used in the mixed effects model. All nitrate peak values were divided by the standard deviation of all nitrate peak values, and the same calculation was used for peak RO responses. The recovery time mixed effects model used response type, disturbance type, mean annual temperature, and mean annual precipitation and used location as a random effect. The response times of RO and nitrate were considered comparable without converting to an effect size. The disturbance type was used instead of the percent of the watershed disturbed as the sole metric of disturbance to reduce redundant information in variables.

3. Results

3.1. Comparison of Methods

RO recovery time estimated by three methods (Methods 2–4) was compared to the estimated RO recovery via paired catchment analysis for catchments with a reference. The recovery times determined by the nonpaired methods were similar to those found by the paired method. The standard error of RO recovery time estimated using the predisturbance RO versus P relationship (Method 2) was ±1.6 years ($R^2 = 0.63$, $n = 8$, $p = 0.02$). The standard error of RO recovery time estimated using the predisturbance runoff ratio (Method 3) was ±0.7 years ($R^2 = 0.89$, $n = 10$, $p < 0.01$). The standard error of RO recovery time estimate using the predisturbance annual loss (Method 4) was ±1.5 years ($R^2 = 0.60$, $n = 10$, $p < 0.01$). Based on these results, sites with limited predisturbance data were analyzed using the runoff ratio method (Method 3) since it was more highly correlated with the paired catchment estimates.

We performed a similar analysis of our two nitrate recovery time estimation methods (Methods N1 and N2) and found similar results. When comparing the nitrate recovery time estimated with predisturbance data and those based on identifying a stable period after disturbance, the standard error was ±0.2 year ($R^2 = 0.97$,

Table 2
Summary of Maximum Increase, Recovery Time and Peak Time of Runoff and Nitrate Concentration

Site	Max runoff increase (mm)	Runoff effect size	Max NO_3^- increase (μmol/L)	NO_3^- effect size	Runoff recovery time (year)	Nitrate recovery time (year)	Runoff peak time (year)	Nitrate peak time (year)
USA								
Hubbard Brook W2	350	2.69	840	4.39	13	6	1	2
Hubbard Brook W4	152	1.17	103	0.54	5	3	0	−2
Hubbard Brook W5	138	1.06	236	1.23	3	4	1	1
Marcell S4	81	0.62			4		0	2
Marcell S6	55	0.42	11	0.057	2	2	0	0
Fewnow WS1	119	0.91			5		0	
Fewnow WS2	429	3.29	63	0.33	3	5	0	2
Fewnow WS3	268	2.06			11		0	4
Fewnow WS5	499	3.83	16	0.084	4	5	2	2
Fewnow WS6	279	2.14	155	0.81	20	16	0	2
Fewnow WS7	254	1.95	133	0.69	20	10	0	4
North Fork Dry Run			61	0.32		16	3	0
White Oak Run	278	2.13	56	0.29	4	9	0	1
Paine Run			33	0.17		9		
Piney River			43	0.22		11		
Staunton River			9	0.047		8		
Japan								
Fukuroyamasawa	301	2.31	168	0.88	>14	6	12	1
Kiryu M			76	0.4		10	−2	3
Oyasan			52	0.27		9	6	4
Teshio	328	2.52	5	0.026	>8	>8	0	4

$p < 0.01$, $n = 5$). In addition, when comparing recovery time when volume-weighted nitrate concentration and absolute value of nitrate export are used, the standard error was ±0.16 year ($R^2 = 0.90$, $p < 0.01$, $n = 15$). This result suggests that the difference arising from the two methods and two indicators (volume-weighted nitrate concentration and nitrate export) is small, and thus, the nitrate recovery times calculated by the two methods can be considered equivalent for the purposes of our study.

3.2. Runoff and Nitrate Response

Fourteen of 20 watersheds showed an RO response to forest disturbance, and 6 watersheds did not. An RO response was detected in all clearcut catchments and the partially cut catchments with a disturbance greater than 75% (Tables 1 and 2). In most natural disturbance sites (pine wilt disease and gypsy moth infestation), a clear RO response could not be detected. The maximum change in annual runoff was 499 mm. In clearcut catchments, the maximum additional runoff ranged from 119 to 350 mm. The timing of the maximum runoff deviation postdisturbance was 1.4 years on average.

A response of stream water nitrate concentration to forest disturbance was detected in all catchments (Table 2). The increase in nitrate concentration ranged from 5 to 840 μmol/L. In Hubbard Brook WS2, the increase in nitrate concentration was highest (840 μmol/L). The peak concentration generally occurred within 2 years after the disturbance (mean delay was 1.8 years), but in the Teshio and Oyasan catchments, the peaks were delayed by 4 years.

The magnitude of the peak response to disturbance was different between runoff and nitrate (Table 3), with runoff showing greater sensitivity (t value = 4.2, $p = 0.0015$). The magnitude of the response was also significantly, positively related to mean annual precipitation (coefficient = 0.004, t value = 2.5, $p = 0.026$). The response was not sensitive to treatment type or mean annual temperature.

Table 3
Summary of the Linear Mixed Effects Model for the Magnitude of the Response to Forest Disturbance and Recovery Time

	Response magnitude		Recovery time	
Variable	DF	p	DF	p
Response type	10	0.002	8	0.86
Disturbance type	14	0.2	13	<0.001
Mean annual temperature (°C)	14	0.21	13	0.03
Mean annual precipitation (mm/year)	14	0.026	13	0.18

Note. The effect of the response type (runoff and nitrate), disturbance type (harvesting type or natural defoliation), and mean annual temperature and precipitation were tested. DF = degrees of freedom.

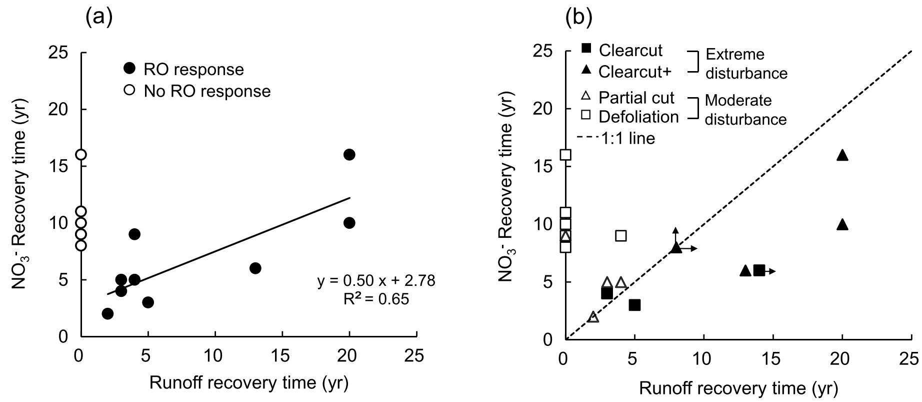

Figure 2. Relationship between runoff and nitrate recovery times when catchments were divided in RO response sites and no RO response sites (a), when divided into clearcut, clearcut+, partial cut, and defoliation sites, and catchments with no runoff and nitrate recovery were included (b). Solid line in Figure 2a represents regression line of catchments with RO response. Arrow in Figure 2b shows that recovery times are larger than the values in the direction of the arrow. Black square (■) and black triangle (▲) indicate clearcut and clearcut plus (herbicide and additional treatment) sites, respectively. These were defined as extreme disturbance. White triangle (△) and white square (□) indicate partial cut and defoliation sites, respectively. These were defined as moderate disturbance. RO = annual runoff.

3.3. Runoff and Nitrate Recovery Time

Runoff recovery time ranged from 0 to 20 years. Two watersheds did not show an RO recovery during our record: Fukuroyamasawa was still recovering after more than 14 years and Teshio catchment after more than 8 years. For analysis of RO recovery time, 12 sites, not including Fukuroyamasawa and Teshio, were used.

Nitrate had recovered in all watersheds excluding one basin (Teshio), which had not recovered after 8 years (Table 2). Thus, our analysis of nitrate recovery time only includes data from 16 sites, excluding Teshio, Marcell S4, Fernow WS1, and WS 3.

The RO and nitrate recovery times were not statistically different. Recovery times were sensitive to disturbance type and mean annual temperature (Table 3). The sites with cut plus additional treatment (Cut+) had the longest recovery times (t value = 4.1, p = 0.001), and other treatments did not have significant effect on recovery time. A positive relationship was observed between the recovery time and mean annual temperature (coefficient = 1.5, t value = 2.4, p = 0.03). Recovery time was not related to mean annual precipitation.

3.4. Relationship Between RO and Nitrate Recovery Times

Including all the data from all the catchments, no significant linear relationship was found between RO and nitrate recovery times (r^2 = 0.05, p = 0.23, n = 15). However, at catchments where RO responses were observed, the relationship between RO and nitrate recovery times was linear (r^2 = 0.65, p = 0.005, n = 9, Figure 2a).

Moreover, RO recovery times and nitrate recovery times were different (Figure 2b). They differed among the moderate disturbance (Defoliation and Partial cut) and extreme disturbances (Cut and Cut+). For moderate disturbance, nitrate recovery times were longer than RO recovery times (i.e., above the 1:1 line). However, for extreme disturbance, the RO recovery times were longer than nitrate recovery times (i.e., below the 1:1 line). In the Fukuroyamasawa watershed, RO recovery was more than 14 years, and nitrate recovery time was 6 years. These data show that nitrate recovery was shorter than RO recovery time.

4. Discussion

4.1. Stream RO and Nitrate Response

Our detection of hydrological change and magnitudes highlights the fact that the disturbance type plays a significant role in RO response. Previous research has shown that disturbances exceeding 20% or greater had a detectable response (Bosch & Hewlett, 1982). No RO change was observed in those with disturbance areas less than 75%, perhaps owing to the limited number of partial clearcuts considered in this analysis. Natural disturbances due to pests and pathogens, which tended to have intermediate spatial coverage within catchments, only showed an RO response in one of six catchments, which is consistent with Helvey and Tiedemann (1978). These responses may be due to the role of trees in enhancing interception or effects of surface roughness and microclimate on evaporation and transpiration (e.g., Biederman et al., 2014). Unlike forest harvesting, the degree of decreased interception and transpiration may be smaller following insect damage and disease because some leaves and stems remain intact. The influence of metabolically impaired trees on catchment hydrology requires additional study. Also, there may be a compensatory effect, such that the transpiration rates of unaffected species increase due to enhanced access to soil water and light. In a snow dominated site, interception and transpiration decreased after forest disturbance, while snow water equivalent increased and the soil water pool and surface evaporation amount increased (Biederman et al., 2015; Penn et al., 2016). In this study, snow-dominated sites are Hubbard Brook, Marcell, and Teshio. The compensatory effect was not found clearly in these areas, but the threshold of snow required to affect the hydrological response following forest disturbance is a subject for future study.

Contrary to RO, nitrate concentration always reacted to disturbances. A stream water nitrate concentration change was detected in each catchment, raising questions about why nitrate dynamics would be more sensitive to disturbance than RO in these catchments. The lack of RO change and the high sensitivity of stream water nitrate in defoliated sites stands out in this comparison. Additional evidence for this pattern is provided by other natural disturbance effects on forest canopies: an ice storm in 1998 at Hubbard Brook (Houlton et al., 2003) and hurricane damage at Luquillo (McDowell et al., 2013). These events damaged the forest canopy, with no evident RO response (Green et al., 2013) but a strong increase in stream water nitrate (Bernhardt et al., 2003).

The difference in the sensitivity of the responses may be due to the differences in the rate of physical compensation mechanisms compared to biological ones. After a forest is disturbed by harvesting/clearcutting, transpiration and interception are reduced, but these fluxes can be almost immediately compensated to a certain extent by soil evaporation (Vertessy et al., 1996), as soil temperatures often increase with canopy removal. The disruption of N uptake by vegetation resulting from forest disturbance may not be quickly compensated for by other biological mechanisms. Denitrification may increase due to wetter and warmer soils in deforested catchments, which may be exacerbated if dissolved organic carbon concentrations increase (Aber et al., 2002; Vitousek et al., 1979). Also, N storage by microbial immobilization and incorporation into the soil organic matter pool likely plays a compensatory role in the loss of N uptake (Vitousek et al., 1982). The compensatory mechanisms in the N cycle are biologically driven, thus likely to take more time to develop than the more physically driven hydrologic mechanisms such as increased soil evaporation and soil storage. Without quick compensation for the loss of N uptake by vegetation, nitrate would be more available in soil water to leach into nearby streams.

While nitrate responses were detected in each case, the degree of response to disturbance type and magnitude was not statistically significant. This may be due to large intersite differences in catchment N cycles, causing the amount of N released after disturbance to vary more based on site than by the external forcing.

4.2. Runoff and Nitrate Recovery Time

The RO recovery times we observed were consistent with previous studies. Annual RO recovery has previously been documented as typically being 3 to 20 years (Brown et al., 2005; Hornbeck et al., 1993; Moore & Wondzell, 2005) but up to a few decades in some snow-dominated catchments (Hicks et al., 1991; Troendle & King, 1985) and managed (e.g., with herbicide) sites (Brown et al., 2005). This research also shows that additional management such as herbicide use makes hydrological recovery time longer.

Our data suggest that RO recovery time for annual streamflow was more related to disturbance than to temperature and precipitation. More sites are needed to confirm this, particularly across more mesic forested areas. If true, the insensitivity of RO recovery to climate variables suggests that streamflow response (defined by recovery time) may be quite resilient to climate change in temperate forests.

Smaller disturbances produced slightly longer nitrate recovery times, and defoliation by insects/disease produced longer recovery times than forest harvesting. It should be noted that other areas have not shown stream water nitrate responses after disturbance of beetle mortality in drier conditions (Rhoades et al., 2013). The reasons for slower recovery after defoliation and less intense harvesting may have to do with the rapid recovery of early successional trees. Large clearings may provide optimal growth conditions. In lower-intensity disturbances, the growth of young trees is limited to gaps where light limitation may be a factor (Lovett et al., 2002). The slow progression of disturbance over several years as the disease and insect damage progress may also be important because it takes time for diseases and herbivores to damage and kill trees.

The positive relationship between mean temperature and recovery time was unexpected and raises questions about its robustness, the potential reasons for the relationship, and the implications. One concern about the robustness of the relationship was that most of the high temperature sites were also sites with herbivore defoliation disturbances (Shenandoah National Park and Kiryu). Generally, warmer climate sites have more insects, so the potential for herbivore defoliation may be high. Warmer climate may produce longer nitrate recovery times, which would likely have implications for N budgets and forest ecosystem functions. Ultimately, more studies across a broader suite of geographies and climates are needed in future syntheses. Additionally, inclusion of more catchment disturbance studies may be needed to validate a relationship between recovery time and temperature.

4.3. Relationship Between Stream Runoff and Nitrate Recovery Times

A relationship between RO and nitrate recovery times (Figure 2a) may reflect a common mechanism. Factors related to RO recovery time are transpiration, ground evaporation, and canopy evaporation (Bruijnzeel, 2004; Verry, 1976; Vertessy et al., 1996). Runoff recovery time is controlled by both physical elements (e.g., weather and canopy roughness) and biological elements (e.g., transpiration and growth rate of trees). Tree height alters meteorological factors and the canopy structure, which in turn alter the amount of evaporation from soil, the canopy, and stomata (Haydon et al., 1996; Murakami et al., 2000; Vertessy et al., 2001). While, nitrogen recovery mechanisms are dominated by biological processes such as uptake of vegetation and processing by microbes (Vitousek & Matson, 1988), hydrological interaction in which feedback on nitrification and denitrification (Paavolainen & Smolander, 1998) and also hydrological transport processes (Riscassi & Scanlon, 2009). The correspondence of RO and nitrate recovery times in this study indicated that forest recovery likely would affect transpiration recovery and nitrate uptake recovery in similar ways.

The comparison between RO and stream water nitrate recovery also demonstrated that the runoff recovery was generally slower than nitrate recovery following extreme disturbance (Figure 2b). We interpret our recovery time data to show a stronger N limitation than water limitation. Further support comes from an analysis using the Budyko framework (Creed et al., 2014) that estimated that PET/*P* (PET is potential evapotranspiration) was smaller than 1 at our sites. This value <1 shows that energy limits evapotranspiration at our sites, not water availability. Our approach of comparing recovery times may help identify dissimilar limitations of N and water on catchment ecosystem function.

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