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Modeling the impacts of wildfire on runoff and pollutant transport from coastal watersheds to the nearshore environment

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ABSTRACT

Wildfire is a common disturbance that can significantly alter vegetation in watersheds and affect the rate of sediment and nutrient transport to adjacent nearshore oceanic environments. Changes in runoff resulting from heterogeneous wildfire effects are not well-understood due to both limitations in the field measurement of runoff and temporally-limited spatial data available to parameterize runoff models. We apply replicable, scalable methods for modeling wildfire impacts on sediment and nonpoint source pollutant export into the nearshore environment, and assess relationships between wildfire severity and runoff. Nonpoint source pollutants were modeled using a GIS-based empirical deterministic model parameterized with multi-year land cover data to quantify fire-induced increases in transport to the nearshore environment. Results indicate post-fire concentration increases in phosphorus by 161 percent, sediments by 350 percent and total suspended solids (TSS) by 53 percent above pre-fire years. Higher wildfire severity was associated with the greater increase in exports of pollutants and sediment to the nearshore environment, primarily resulting from the conversion of forest and shrubland to grassland. This suggests that increasing wildfire severity with climate change will increase potential negative impacts to adjacent marine ecosystems. The approach used is replicable and can be utilized to assess the effects of other types of land cover change at landscape scales. It also provides a planning and prioritization framework for management activities associated with wildfire, including suppression, thinning, and post-fire rehabilitation, allowing for quantification of potential negative impacts to the nearshore environment in coastal basins.

1. Introduction

Wildfire is an integral natural disturbance in many ecosystems. Anthropogenic climate change, however, is predicted to increase fire activity progressing through the 21st century (Abatzoglou and Kolden, 2011; Littell et al., 2009), creating disturbance patterns that may alter ecosystems in unprecedented ways. Wildfire is arguably the most common ecological disturbance in Mediterranean ecosystems of coastal California watersheds draining into the Pacific ocean (Keeley and Zedler, 2009), where large coastal wildfire events have been shown to negatively impact sea otter (*Enhydra lutris*) immune response (Bowen et al., 2014; Venn-Watson et al., 2013). As a key marine mammal predator, the sea otter is an indicator of nearshore ecosystem health and is listed in California as "Threatened" under the Endangered Species Act and protected under the Marine Mammal Protection Act (U.S. Fish and Wildlife Service, 2014). Despite this protection, the California sea otter population is recovering at a lower-than-expected rate, leading to queries seeking to identify factors impeding population growth rates (Johnson et al., 2009). Inputs to the sea otter's nearshore habitat from terrestrial watersheds, such as toxins, nutrients, and pollutants, have been shown to negatively affect sea otter health (Conrad et al., 2005; Johnson et al., 2009; Miller et al., 2010), but prior studies have focused primarily on pathogens (Johnson et al., 2009) or large, anthropogenic spill events like the Exxon Valdez oil spill of 1989 (Bodkin et al., 2002). This focus overlooks the contributions of ecological disturbance events like wildfire, in part because the pathways for transport of pollutants have not previously been explicitly identified or modeled.

Wildfire has the potential to alter terrestrial inputs to the adjacent nearshore environment by significantly altering the condition of soil and vegetation affecting infiltration and transport of

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nutrients and metals (Stein et al., 2012) and increasing erosion and sediment yields (Warrick et al., 2012; Moody et al., 2013). Wildfire increases the amount of phosphorus and nitrogen in streams (Spencer and Hauer, 1991; Coombs and Melack, 2013), even many years after the fire (Hauer and Spencer, 1998), as well as sediment and nutrient loads (Stein et al., 2012; Warrick et al., 2012), suggesting that large wildfires should also impact the nearshore environment as these nutrients and sediments then eventually drain into the ocean. The effects of wildfire on the increasing transport of sediment and nutrients have been shown to adversely affect freshwater aquatic ecosystems (Gresswell, 1999; Spencer et al., 2003), however, the effect of these increases in marine ecosystems has not been well documented.

The most common approach to measuring the specific effects of wildfire on runoff, sediment transport and nutrient loading has been through *in-situ* stream sampling (e.g., Stein et al., 2012). However, this approach severely limits the ability to understand the cumulative impacts of both the wildfire and the pre- and post-fire management actions (e.g., burned area rehabilitation efforts) on the nearshore ecosystem, for two primary reasons. First, wildfires often burn only portions of watersheds, and burn with variable severity across watersheds, but *in-situ* sampling does not account for this spatial heterogeneity (Shakesby and Doerr, 2006). Additionally, the results from studies collecting *in-situ* measurements are not transferable or easily comparable to other fires or watersheds across time and space due to differences in size, composition, and management impacts that are difficult to tease out from the single sample value. To our knowledge, no prior studies have attempted to model the spatially explicit impacts of a wildland fire event on runoff into the nearshore environment using a replicable, scalable, watershed approach.

Prediction of post-fire effects commonly has focused on runoff and erosion rates and relied on a variety of physically and empirically based models, spatially distributed models, and professional judgment (Larsen and MacDonald, 2007). Commonly used post-fire runoff models such as ERMIT (Robichaud et al., 2007), RUSLE (Renard and Foster, 1991), and Disturbed WEPP (Elliot and Hall, 2010) include land cover as an input, but do not maintain the spatially explicit information (*but see* Renschler, 2003), limiting the ability to link spatially variable wildfire effects and land management actions to model outputs. These models have also produced a wide range of runoff estimates, making comparisons difficult (Robichaud et al., 2000), and few have been validated in post-fire environments (Larsen and MacDonald, 2007). Among models that have been validated, results have demonstrated that the amount of vegetation cover post-fire has a strong impact on erosion rates (De Dios Benavides-Solorio and MacDonald, 2005).

A replicable approach to quantify and compare spatially explicit impacts of wildfire on nonpoint source pollutants must utilize standardized model inputs that reflect the spatial and temporal variability of wildfire effects and follow a clear framework for how fire affects model inputs. The primary input affected by wildfire at landscape scales is land cover, which is often classified from remotely sensed data and represents both vegetation and human development. Widely available data such as the National Land Cover Dataset (NLCD) (U.S. Geological Survey, 2014) are frequently used for modeling efforts that require land cover, but can be limiting because they are produced at infrequent intervals due to the challenges of acquiring adequate remotely sensed data and the intense effort required to produce continental-scale classifications. For example, the most recent NLCD data when this research was conducted was produced in 2006 and as a result does not account for subsequent years of land cover transition, including two large wildfires that burned on the Big Sur coast in 2008. Incorporating these land cover disturbances is essential to accurately modeling changes in transport of nonpoint sources of nutrients and sediments, and relies upon utilizing information about how the spatially heterogeneous severity of the fire impacts land cover.

As a result of wildfire, vegetation communities continue along established succession pathways or undergo type conversions from one community to another along alternative pathways depending on fire frequency and severity (Larson et al., 2013). The severity of a fire, often described as 'burn severity', or the magnitude of change in the post fire environment (Key and Benson, 2006), impacts vegetation succession and pattern (Larson and Churchill, 2012; Lutz et al., 2013), vegetation composition and structure (Lutz et al., 2012; Kane et al., 2013; Cansler and McKenzie, 2014) and therefore the potential for increased erosion and flooding (Robichaud et al., 2000). Understanding burn severity across landscapes and the resulting changes within these landscapes is especially important when considering effects occurring in coupled ecosystems like the nearshore environment and its adjacent terrestrial watersheds. Much research has been focused on characterizing burn severity through remotely sensed data (Van Wagtendonk et al., 2004; Key and Benson, 2006; Cansler and McKenzie, 2012; Kolden and Rogan, 2013; Kane et al., 2014) and looking at changes and trends in burn severity (Kolden et al., 2012; Miller and Safford, 2012). While several longitudinal studies have monitored wildfire influences on vegetation succession at the plot scale (Callaway and Davis, 1993; Santana et al., 2012; Halpern and Lutz, 2013), comparatively fewer studies have looked at relationships between burn severity and vegetation at landscape scales. These have mostly focused on characterizing pre-fire vegetation contributions to post-fire severity (Kolden and Abatzoglou, 2012; Birch et al., 2014) rather than linking severity to post-fire transitions, in part due to the relatively recent development of representative metrics that allow burn severity to be mapped.

Quantifying the relative impacts of wildfire on land cover change and subsequent runoff is critical to understanding how terrestrial disturbance and change can impact threatened species in the nearshore environment. While most studies surrounding the limited growth of the sea otter population in central California have focused on anthropogenic inputs (Conrad et al., 2005; Dowd et al., 2008; Johnson et al., 2009), the Big Sur population of sea otters is comparatively isolated and protected from anthropogenic inputs, but still limited in terms of population growth. We hypothesized that large wildfires have similar detrimental effects as anthropogenic inputs to the nearshore environment due to fire-altered land cover, and that the use of a spatially explicit runoff model would demonstrate the relationship between greater burn severity and higher runoff following wildfire. Here, we demonstrate an approach to assess the sensitivity of nonpoint source pollutants and runoff into the nearshore environment to wildfire-induced changes in land cover. Our objectives were: 1) to characterize the effect of burn severity on land cover within the study area, 2) to model nonpoint source pollutants utilizing a multi-year land cover time series that incorporates wildfire effects, and 3) to quantify the changes in modeled nonpoint source pollutants to the nearshore environment resulting from the fire effects.

2. Methods

2.1. Study area

The study area is located on the central California coast in an area formed by twelve adjacent watersheds covering 87,638 ha and draining a portion of the Santa Lucia Range in the northern portion of the Los Padres National Forest (Fig. 1). The Santa Lucia Range rises steeply from sea level to just below 1800 m within a few km from the coast, and experiences a Mediterranean climate, with fire

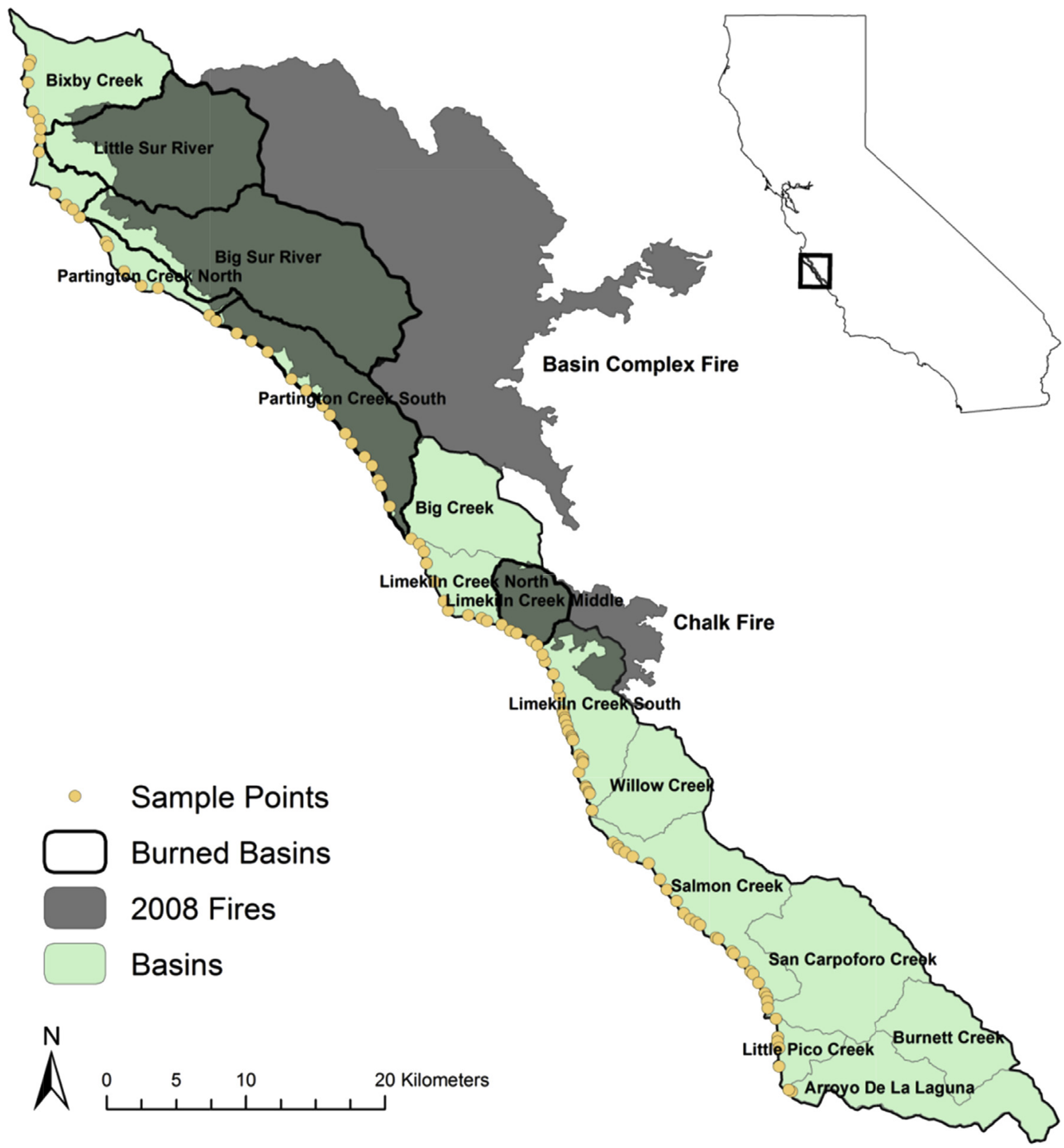

Fig. 1. Basins and sample points in the Big Sur region of coastal central California (where sample points are defined as the point just upstream from where the water course enters the ocean). Burned basins are in bold.

season typically lasting from June to November (Greenlee and Langenheim, 1990). Precipitation is dependent on elevation ranging from 65 cm near the coast to over 130 cm at ridge top (Davis et al., 2010). Average temperature generally increases from north to south and with distance from the coast (Davis and Borchert, 2006). These weather and elevation gradients create a highly diverse ecosystem which has been identified as a global biodiversity "hotspot" (Myers et al., 2000), including three ecological zones within the study area. These zones are comprised of grasslands, coastal sage scrub, chaparral, oak forests, mixed broadleaf evergreen forest, and coniferous forests (Davis and Borchert, 2006).

In the Big Sur region, the majority of area burned is from large infrequent fires controlled primarily by extreme weather with return intervals estimated to be 75 years on average (Davis and Borchert, 2006). Prior to 2008, the most recent fires to burn through portions of the study area were the 1977 Marble Cone Fire (72,500 ha) and the 1999 Kirk Complex (35,100 ha). In 2008 two large fires burned approximately 33,038 ha, or 42 percent of the study area. The Basin Complex Fire burned from June 21 to July 27, 2008 and the Chalk Fire burned from September 27 to October 30. Both fires were contained before the onset of the winter rainy season (Davis and Borchert, 2006).

2.2. Fire effects

To characterize the effects of the 2008 wildfires on land cover, classifications of land cover and burn severity derived from remotely sensed data were explored. A land cover dataset for each year from 2005 to 2012 was developed in order to capture the effects of the 2008 fires; the overall classification accuracy ranged from 75 to 90 percent for the eight years of land cover maps produced (Morrison and Kolden, 2014). Land cover was classified into forest, shrub, or grass. These three classes are the dominant vegetation types across the study accounting for 97 percent of all land cover. Because bare ground was such a small percentage of the study area (<0.01 percent) it was not specifically classified and was primarily classified as grass because of its spectral similarity to senesced grass in the study. Due to the date of image acquisition (August 1, 2008) and the timing of the early summer Basin Complex Fire (June) versus the late ignition of the Chalk Fire (September), fire effects from the Basin Complex are recorded in the 2008 land cover data approximately one month after the fire burned while fire effects from the Chalk Fire are recorded approximately one year post fire in the 2009 land cover data. Land cover changes for each class were quantified by plotting the change in percent cover over time of both burned and unburned areas.

Landscape-scale burn severity is commonly represented by a spectral index called the differenced Normalized Burn Ratio (dNBR) (Key and Benson, 2006). The Normalized Burn Ratio (NBR) is calculated from atmospherically-corrected at-surface reflectance in the near-infrared (NIR) and short-wave infrared (SWIR) bands:

$$\mathrm{NBR} = \frac{(\mathrm{NIR} - \mathrm{SWIR})}{(\mathrm{NIR} + \mathrm{SWIR})}$$

The differenced equation, dNBR, is produced by subtracting the post-fire NBR image from the pre-fire NBR, creating an index representing a magnitude of change (Key and Benson, 2006). Changes in NIR wavelengths indicate a change in green vegetation and biomass (Jensen, 1983) whereas SWIR wavelengths are documented to have sensitivity to soil and plant moisture (Jensen, 2007) as well as burnt vegetation, ash, and exposed soil (Smith et al., 2005). The dNBR for the Basin and Chalk fires was calculated from Landsat 5 Thematic Mapper scenes from May 13, 2008 and May 16, 2009, path/row 43/35.

Four metrics were calculated to represent the effects of fire for each basin (Table 1). First, percent high burn severity (HS) was calculated from the dNBR using a threshold of 367, which was identified by Miller and Thode (2007) as being correlated to field measures in a variety of vegetation types in the Sierra Nevada in California having high to complete vegetation morality. Second, the Severity Metric (SM; Lutz et al., 2011) was calculated based on the dNBR. Lutz et al. (2011) developed the Severity Metric to represent burn severity continuously. Grouping burn severity or using a threshold to determine discrete classes is necessary as a means to communicate the effects of fire and in accounting for differences in scale and diverse methods used to measure burn severity for different fires (Miller and Thode, 2007). However, separating burn severity into discrete categories can be problematic when classes differ between studies (Miller and Thode, 2007), and information can be lost in the process (Lutz et al., 2011).

Finally, two transitions in land cover were calculated: 1) the percent of pixels that transitioned from forest to shrub or from forest to grass (characterized as 'Forest Loss') and 2) the percent of pixels that transitioned from shrub to grass ('Shrub Loss'). These two vegetation transition metrics were calculated for two temporal periods for each watershed, from 2006 to 2008 and from 2006 to 2009, to capture the effects of both fires. For consistency, 2006 was chosen as the pre-fire year instead of 2007 because the study area received only 50 percent of average precipitation in 2007 (California Department of Water Resources, 2010), and the subsequent drought stress effects on vegetation had a noticeable effect of reducing the accuracy of the 2007 land cover classification (Morrison and Kolden, 2014). Because not all burned basins were 100 percent burned, the majority of remaining pixels not categorized as Forest Loss or Shrub Loss did not transition but remained the same between the years. The remainder of pixel transitions (<2 percent) was considered to be background noise as changes in land cover classes can be a result of classification accuracy and not an ecological process (Foody, 2002).

Table 1
Basins within the study area from north to south and fire impacts including fire name, percent area burned, percent high severity (%HS), and Severity Metric (SM). Basins in bold are considered to be the burned basins for analysis and were >75% burned.

Basin	Area (km^2)	Fire	%Area burned	%High severity (HS)	SM
Bixby Creek	66.2	Basin	3.5	1.8	0.3772
Little Sur River	102.9	Basin	87.1	39.1	0.3954
Big Sur River	148.5	Basin	91.9	38.0	0.3783
Partington Creek North	16.5	Basin	9.5	0.2	0.178
Partington Creek South	65.1	Basin	92.5	29.5	0.3283
Big Creek	55.7	Basin/ Chalk	1.0	0.2	0.2892
Limekiln Creek North	22.0	Chalk	1.4	0.3	0.3209
Limekiln Creek Middle	23.7	Chalk	98.8	19.4	0.2903
Limekiln Creek South	52.2	Chalk	21.8	7.4	0.3545
Willow Creek	41.0		0	0	0
Salmon Creek	70.3		0	0	0
San Carpoforo Creek	90.6		0	0	0
Little Pico Creek	14.2		0	0	0
Arroyo De La Laguna/ Burnett Creek	107.9		0	0	0

2.3. Hydrologic effects

Annual accumulation of nonpoint source pollutants were modeled using the Open Nonpoint Source Pollution and Erosion Comparison Tool (OpenNSPECT) (Eslinger et al., 2012). OpenNSPECT is an open-source Geographic Information Systems (GIS)-based tool that models pollutants and erosion loads delivered to coastal watersheds (NOAA, 2012). OpenNSPECT relies on the relationship between land cover, nonpoint source pollutants, and erosion to estimate accumulations from overland flow within a watershed (NOAA, 2012). Inputs to the model include elevation, land cover, rainfall, and soil data, R-factor, and pollutant coefficients.

Runoff is modeled as the basis for OpenNSPECT processes using methods developed by the National Resource Conservation Service (NRCS) (NRCS, 1986). It relies on the inputs of rainfall, elevation, land cover and soil and produces an accumulated runoff grid output of volume in liters. Pollutant concentrations are estimated using land cover as an indirect means by which coefficients determined by water quality standards represent the contribution of each land cover class to overall pollutant load during a precipitation event (NOAA, 2012). Five default pollutants are measured: nitrogen, phosphorus, total suspended solids (TSS), lead, and zinc. Using pollutant coefficient values – which represent an average concentration (mg/L) for each cell's land cover – type and a flow direction grid from the DEM, a pollutant mass accumulation and accumulated pollutant concentration is calculated (Appendix A).

Rates of erosion and sediment loads are estimated with a modified version of the widely used revised universal soil loss equation (RUSLE) (Renard and Foster, 1991):

$$A = R*K*L*S*C*P$$

where A is the average annual soil loss, R is a rainfall/runoff erosivity factor, K is a soil erosivity factor, L is the length-slope factor, S is the slope steepness factor, C is the cover management factor, and P, the support practice factor, is not included in the OpenNSPECT calculation (NOAA, 2012). RUSLE uses these factors to represent the processes of infiltration, overland flow, particle detachment, and sediment transport (Larsen and MacDonald, 2007). User inputs needed to model sediment yields include elevation, land cover, soils, and the rainfall/runoff erosivity factor (R factor). The output is a gridded annual accumulation of sediment yield.

OpenNSPECT is dependent upon land cover for predicting runoff, pollutants, and erosion; therefore, changes in land cover inputs function primarily as a model sensitivity study. Model outputs based on changes in land cover were predicted by running OpenNSPECT for each year from 2005 to 2012. To accomplish this, each year an updated land cover dataset was used which incorporates the effects of fire in 2008. The years 2005–2007 were years without fire or pre-fire years, 2008 included effects of the Basin Complex Fire, and 2009–2012 included both the Basin Complex and Chalk fires in post-fire years.

Hydrologic effects were measured at the basin level and were also aggregated by total burned and unburned. Sub-basins that were predominantly burned or unburned were delineated in order to more clearly separate burned and unburned areas. Partington Creek basin was divided into one unburned and one burned basin, and Limekiln Creek was divided into two unburned basins and one burned basin. All accumulation grids were overlapped with stream data and only channels that matched actual perennial and intermittent streams were considered for sampling. For all output grids, between 1 and 24 sample points or "pour points" were identified to quantify outputs for a given basin. These pour points represent the locations where streams drain into the ocean for a given basin, with the number of pour points for a basin essentially representing the number of well-defined streams (or sub-basins) that drain into the ocean for that basin. Pour points were identified for 14 of the 15 basins (Fig. 1); the Brunette Creek basin drains into Arroyo De La Laguna basin before reaching the ocean, and therefore the two were merged and points were only taken from the pour point of Arroyo De La Laguna basin. Annual accumulation of loads and concentrations were gathered at each pour point. For comparing burned to unburned area, pour points were summed from basins where >75 percent of the area burned (4 basins) for the burned group; the remaining 10 basins were considered unburned. Percent change values were calculated as annual averages based on the years of interest for each question.

2.4. Analysis

Impact of fire on hydrologic responses was visualized by comparing the percent change in modeled nonpoint source pollutants concentrations from burned and unburned basins from a pre-fire baseline (2005–2007) from all outputs from OpenNSPECT. In 2008, only the basins burned by the Basin Complex Fire were used to calculate the pre-fire baseline and the percent change in export. For all other years, all burned basins were included in the pre-fire baseline and the percent change in nonpoint source pollutants.

Spearman Rank correlation was used to explore the relationship between the changes in nonpoint source pollutants and the fire metrics. Percent changes in nonpoint source pollutant concentration transport per basin were compared to each fire metric for the two time periods (2006–2008 and 2006–2009). Percent change in basin nonpoint source pollutants for 2006–2008 and from 2006 to 2009 were correlated to Severity Metric (SM), proportion High Severity (HS), and Forest Loss and Shrub Loss. Spearman Rank was used to account for the small sample size of basins ($n = 14$) and the non-normal distributions (Wilks, 1995) Significance was measured at the $p < 0.05$ level.

3. Results

3.1. Land cover

Within fire perimeters, average pre fire (2005–2008) land cover proportions were 52 percent forest, 43 percent shrub and 5 percent grass (Fig. 2). In 2008, after the Basin Complex Fire, grass cover increased to 46 percent as forest decreased to 11 percent with shrub maintaining at 43 percent. In 2009, approximately one year after both the Basin Complex and Chalk fires, forest decreases further to 8 percent of total cover, shrub increases to 76 percent and grass cover decreases to 16 percent. Post-fire land cover proportions (2010–2012) were 28 percent for forest cover, which

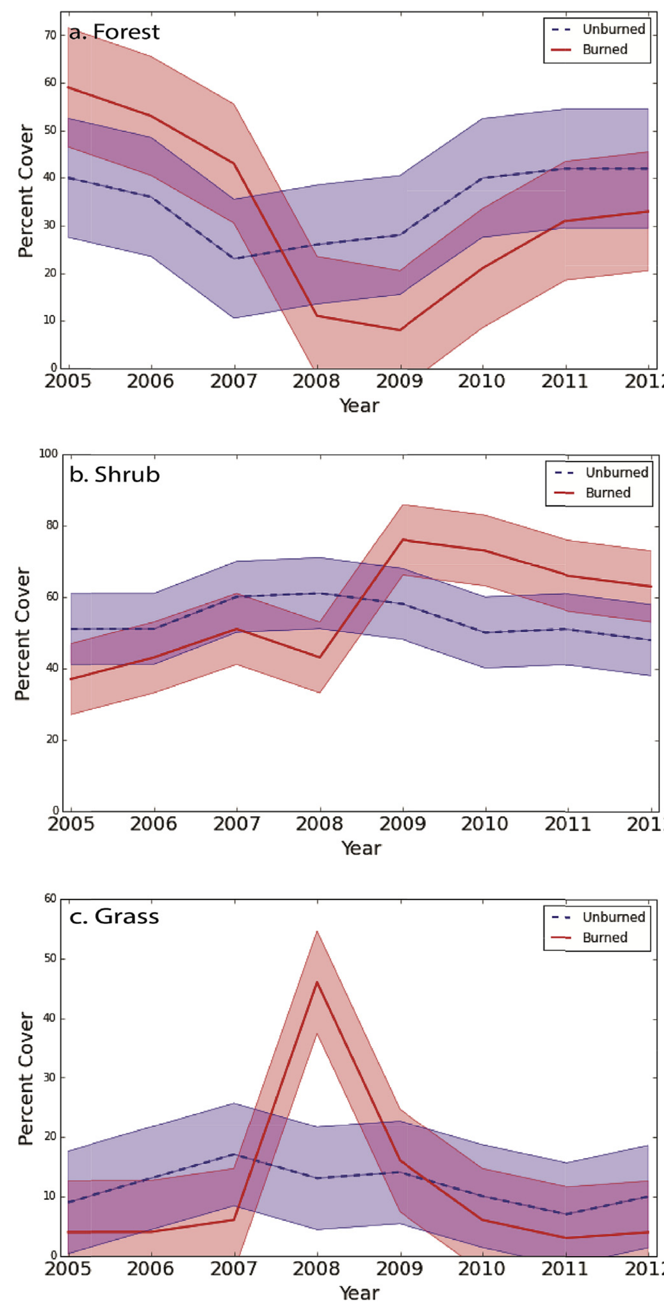

Fig. 2. Burned and unburned land cover transitions in (a) forest, (b) shrub, and (c) grass cover over the study period (2005–2012). Envelopes represent the average error of commission as a percent for each class.

increased slightly each post-fire year but still had less total cover than in pre-fire conditions. Shrub post-fire cover was 67 percent, which decreased each post-fire year but was still higher than pre-fire conditions. Average post-fire grass cover was 4 percent, which was consistent with pre-fire grass cover (percentages do not equal 100 due to rounding).

3.2. Watershed outputs

Modeled annual concentrations of phosphorus (P), Total Suspended Solids (TSS), and sediment summed within burned basins all increased above pre-fire baseline levels during 2008 (Fig. 3) and 2009. There was little change in nitrogen. Relative percent change in 2008 included only the basins affected by the Basin Complex Fire and showed that in 2008 there was 161 percent increase in phosphorus, a 115 percent change in TSS, a 337 percent increase in sediment, and a 26 percent increase in runoff. In 2009, all of the burned basins had a 71 percent increase in phosphorus, a 53 percent increase in TSS, a 109 percent increase in sediment, and a 4 percent decrease in runoff over pre-fire baseline levels. For phosphorus, TSS, and sediment, pre- and post-fire levels were similar to those of unburned areas. Runoff, however, was higher in burned areas than in unburned areas pre-fire, but post-fire runoff dropped below that of unburned areas.

3.3. Burn severity and watershed export

Fire metrics were significantly correlated to relative change in nonpoint source pollutants at the $p < 0.01$ level (Table 2). The highest correlations for change across post-fire year 0 (2006–2008) were between Forest Loss and both phosphorus and sediment ($r = 0.89$, $p < 0.001$), and between SM and TSS ($r = 0.89$, $p < 0.001$). Overall, all of the fire metrics were strongly correlated with all of the relative changes in export, except for Forest Loss and runoff, which was not significant. The highest change across the post-fire year 1 (2006–2009) period was between HS and phosphorus. For this period, runoff was not significantly correlated with any for the fire metrics. Omitting runoff, overall, for post-fire year 1, HS showed the strongest correlations with changes in export, while Forest Loss has the weakest correlations.

Table 2
Spearman Rank correlations (r) between fire metrics and percent changes in modeled annual concentrations for the two periods of post-fire year 0 (2006–2008) and post-fire year 1 (2006–2009). Fire metrics include proportion High Severity (HS), Severity Metric (SM), Forest Loss (FL), and Shrub Loss (SL). Values in bold are significant at the $p < 0.01$ level.

	ΔPhosphorus	ΔTSS	ΔSediment	ΔRunoff
Post-fire year 0				
HS	**0.82**	**0.83**	**0.75**	**0.72**
SM	**0.75**	**0.89**	**0.76**	**0.75**
FL	**0.89**	**0.84**	**0.89**	0.44
SL	**0.72**	**0.81**	**0.73**	**0.88**
Post-fire year 1				
HS	**0.88**	**0.87**	**0.87**	0.33
SM	**0.82**	**0.81**	**0.85**	0.37
FL	**0.76**	**0.76**	**0.78**	0.09
SL	**0.81**	**0.84**	**0.82**	0.28

4. Discussion

The results reveal that wildfire significantly altered land cover for 2–3 years following the fire year, subsequently resulting in a 2-year period of dramatically increased sediment and nonpoint source pollutant yield into the adjacent nearshore ecosystem. A reduction in forest and shrub cover in 2008 resulted in a large increase of grass cover which is due to the removal of shrub canopy promoting the growth of herbaceous annuals and perennials (Davis and Borchert, 2006; Keeley, 2006a). Keeley et al. (2005) observed that in post-fire California Mediterranean shrublands during the first spring post-fire, approximately 50 percent of post-fire cover

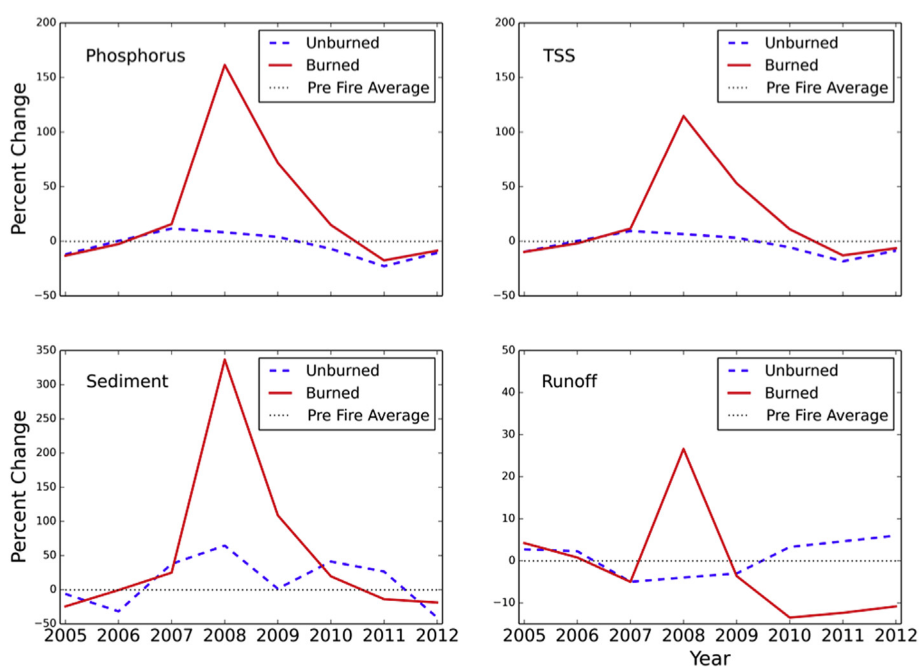

Fig. 3. Percent change in nonpoint source pollutant concentrations based on pre-fire average (2005–2007) for both burned and unburned basins for Phosphorus (top left), Total Suspended Solids (TSS; top right), Sediment (bottom left), and Runoff (bottom right). Within burned areas, 2008 includes only the burned basins and pre-fire baselines affected by the Basin Complex fire.

was composed of herbaceous annuals in the interior sage scrub communities and perennials in the coastal sage scrub.

In 2009, one year after both fires, shrub increases to 76 percent of the land cover. Shrub cover in California Mediterranean ecosystems consists of a variety of species that are obligate resprouters, obligate reseeders and facultative seeding species (Keeley, 2006b). Coastal sage scrub communities are comprised of primarily resprouters, while interior sage scrub is composed of fewer resprouting species and more facultative seeders that rely more on obligate seeding (Keeley, 2006a). Dependent on slope and aspect, chaparral communities are a mixture of obligate seeders on xeric sites while resprouting species occupy more mesic sites (Keeley, 2006a). All shrub species in this ecosystem are shown to regenerate in response to fire (Keeley, 2006b) which is consistent with a large increase in shrub measured in 2009. Keeley et al. (2005) also found that in a five-year study, shrub cover increased in each post-fire year, though shrub growth can vary post-fire (Keeley and Keeley, 1981). After 2009, there was subsequent decrease in shrub cover in every post-fire year (Fig. 2) which could also be attributed to resprouting of coast live oak (Davis and Borchert, 2006) that is classified as shrub initially, or spectral confusion between the shrub and forest classes. These land cover transitions generally align with other studies that have shown that increasing density of vegetation decreases runoff (Nicolau and Solé-Benet, 1996; Garcia-Estringana et al., 2010) and the presence and type of forest greatly reduces erosion (Descroix et al., 2001). In the three post-fire years after 2009, grass cover returned to pre-fire levels, whereas forest cover remained below pre-fire levels and shrub remained above.

OpenNSPECT pollutant coefficients produce highest mass and concentration outputs for grass out of the three cover classes, but are the same for shrub and forest. For some pollutant coefficients such as nitrogen, all three cover classes produce the same mass and volume, making the model insensitive to changes in land cover for certain pollutants. This is why modeled nitrogen did not show any significant changes between years. The increase of grass above pre-fire levels (and subsequent decrease in forest and shrub) in 2008 and to a lesser extent in 2009 is therefore the driving cause of increases in modeled pollutants. By 2012, percent change in nonpoint source pollutant levels is near or below pre-fire levels, though post-fire vegetation proportions differ from pre-fire.

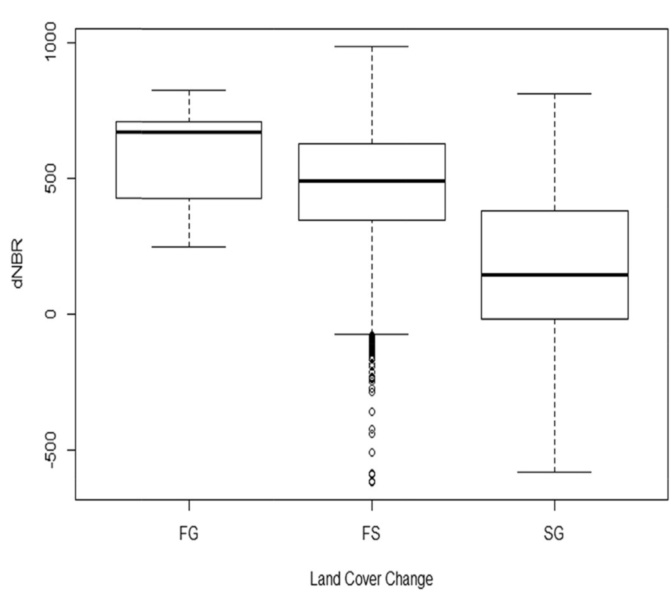

Fig. 4. Distributions of dNBR per land cover transition. FG = forest to grass, FS = forest to shrub, and SG = shrub to grass.

When comparing the changes in nonpoint source pollutants by basin, those with the highest indications of fire severity also showed the largest modeled increases in nonpoint source pollutants. This indicates that increased burn severity is linked to certain land cover transitions. Higher levels of burn severity are associated with greater loss of forest and shrub and increases in grass to produce increases in nonpoint source pollutants, while unburned basins or basins low fire metric values showed less increase in pollutants and greater forest and shrub loss. Major transitions (i.e. from forest to grass) are occurring at the higher values of dNBR (Fig. 4) compared to transitions between forest and shrub and shrub to grass. The severity of a fire is one of the most important influences on post-fire erosion rates in many ecosystems (De Dios Benavides-Solorio and MacDonald, 2005). Likewise, the highest levels of dNBR lead to changes in land cover that produced increased modeled changes in nonpoint source pollutants.

The use of a multi-year land cover dataset helped to explicate the impacts of the 2008 wildfires through changes in land cover and subsequent increases in modeled nonpoint source pollutants to the nearshore ecosystem above non-fire years. These are effects that would not have been observed using only pre-fire land cover data. Various studies using in-situ stream measurements have found increases in watershed transport of nutrients in storm events after a fire. Hauer and Spencer (1998) collected stream nutrient data during a fire from a series of paired watersheds in Montana. Their results for concentrations are 5–10 fold higher for phosphorus and 13–25 fold higher for TSS. In southern California coastal watersheds, Stein et al. (2012) found a 921-fold increase in phosphorus concentration and a two-fold increase in TSS after post-fire storm events. Our modeled annual phosphorus concentrations showed a 161 percent increase compared to pre-fire average and a 114 percent increase in TSS over a pre-fire average. Modeled increases are expected to be lower as they are measured over an annual scale. Studies have also found significant increases in sediment transport during early season storms in southern California (Warrick et al., 2012; Coombs and Melack, 2013) which corresponds to the 350 percent modeled annual increase in sediment concentrations in the fire year.

An additional factor that compounds the difficulty of comparing our results to *in-situ* measurements, but also elicits information that an *in-situ* study cannot easily acquire, was our use of climatological mean annual precipitation, instead of the precipitation for the individual years. There are no weather observing stations in the study area that capture representative rainfall for the whole region, and we specifically desired to control for the effects of precipitation in order to assess sensitivity to land cover change and wildfire effects. This allows us to also interpret our results to quantify the longevity of fire effects on runoff and pollutant transport; as soon as land cover reached pre-fire levels of grass (only 2 years post-fire), model outputs returned to baseline levels. This signal would have been confused if precipitation had not been controlled for, particularly as the years 2007 through 2009 included a droughty period with below normal precipitation (which was likely a primary driver of the fire activity), but diminishes comparability with *in-situ* studies.

Though results agree with studies in finding orders of magnitude increases in nonpoint source pollutants post-fire, modeled concentrations tended to overestimate concentrations of nonpoint source pollutants compared to studies in similar areas (Warrick et al., 2012) or to ecosystem specific water quality standards (US EPA, 2000). Modeled phosphorus concentration in unburned years was about 5 times higher than ambient water quality criteria developed for the southern and central California chaparral and oak woodlands. The cause of this disagreement between concentrations can potentially be attributed to difference between spatial and temporal extents. Often post-fire runoff studies lack spatial and temporal context for reported result (Shakesby and Doerr, 2006). The same is likely true for water quality standards generated at local and regional scales. Studies collecting *in-situ* stream data are also primarily conducted in response to individual storm events, while our modeled data are based on an annual average of precipitation with no extremes; responses are averaged throughout the year.

Along with challenges of comparison with *in-situ* studies, several limitations to our approach are observed. The model is primarily for small and mostly urban watersheds (NOAA, 2012), and there are inaccuracies inherent in each of the data inputs. For land cover data, accuracy is spatially heterogeneous, making it difficult to pinpoint specific locations of errors. Larsen and MacDonald (2007) tested the accuracy of two annual time scale erosion models: the physically based Disturbed WEPP and empirically based RUSLE models to predict post-fire sediment yields in the Colorado Front Range and found that they were poorly correlated to actual sediment yields. Specifically, the RUSLE model, designed to predict long-term annual averages of soil loss for conservation planning and assessment, does not account for sediment in channels and is more appropriate for small areas (Nearing et al., 2005).

True post-fire conditions are not reflected in model processes and inputs, but this is actually an advantage of modeling over *in-situ* studies when seeking an understanding of process. Post-fire runoff is relatively poorly understood (Moody et al., 2013; Stein et al., 2012) especially in California chaparral watersheds (Coombs and Melack, 2013). Land cover is represented as static across the year in the model; however, in post-fire environments there are interannual changes in fire year vegetation. It is likely that during the first storm event after the fire, vegetation cover is sparser but continues to grow throughout the rainy season. Sparser vegetation cover would lead to increased transport of nonpoint source pollutants (Robichaud et al., 2000) especially considering the effects of fire on soils. Severe wildfire produces highly spatially variable hydrophobic soils in California chaparral (Hubbert et al., 2006) which is not accounted for in the soil dataset and various degrees of soil hydrophobicity within high and moderate soil burn severity ratings were produces during the Basin Complex Fire (SEAT, 2008). Also, combustion of plants and other natural materials releases nutrients (Ranalli, 2004) in ways that are distinct from runoff from agricultural or developed land cover and are not accounted for in OpenNSPECT. Wildfire creates an increase in nutrients such as phosphorus and nitrogen primarily through smoke and the deposition of ash through overland flow (Ranalli, 2004).

Despite these limitations and uncertainties, our modeled increases in nonpoint source pollutants correspond with post-fire research by generally showing increases in runoff, sediment, and nutrients. Due to additional source of nutrients not modeled through OpenNSPECT, we believe that this model produces a conservative estimate of the export of nutrients and sediment from these coastal watersheds. A water quality management plan for a watershed in Morro Bay just south of Big Sur indicates that sediment loading is 50 percent higher than the established total maximum daily load (TMDL) and would be even higher in the event of a wildfire in the basin (State of California Central Coast Regional Water Quality Control Board, 2002). Our modeled results indicate a 350 percent increase in sediment yield. An increase of this magnitude violates these established water quality standards from nearby coastal watersheds. Although central California coastlines are relatively unimpaired (Green et al., 2004) nutrient loading and the growth of toxic algae has been documented to cause sea otter mortality in Monterey Bay (Miller et al., 2010). Though we modeled a 161 percent increase in phosphorus, these elevated levels were fleeting, returning to pre-fire levels two years after fire. This may not be enough of an increase to create impaired coastal waters, however, incorporating more representative post-fire parameters, such as soil hydrophobicity, annual precipitation, and additional land cover classes would likely lead to higher levels of increase than modeled.

Climate impacts have shown to be increasing the severity (Miller and Safford, 2012), occurrence (Westerling and Bryant, 2008; Lutz et al., 2009), and area burned of fires across California (Abatzoglou and Kolden, 2013). For weather-driven fire on the central California coast (Moritz, 1997) increasing temperatures could lead to lead to further increases in wildfire activity. Although it is uncertain how precipitation may change regionally across California within the next century (Cayan et al., 2008) the sensitivity of the modeled output to changes in land cover indicates that increases in nonpoint source pollutants to the nearshore are coupled with increased occurrence and severity of wildfire.

A number of coastal California water quality management plans include the marine ecosystem as a “beneficial use” (State of California Central Coast Regional Water Quality Control Board, 2002) while post-fire emergency assessments in coastal watersheds do not (SEAT, 2008; USDA Forest Service, 2010). A wildfire occurring within the in Morro Bay south of Big Sur is considered a situation in which water quality standards (TMDLs) will not be met (State of California Central Coast Regional Water Quality Control Board, 2002). Therefore, it should also be important to consider the nearshore ecosystem as a “value at risk” for fire-prone coastal wildland areas.

5. Conclusion

Understanding the impacts of increases in wildfire on ecosystems is important especially when considering coupled ecosystems of coastal watersheds and the nearshore environment. Recent research has shown the effects of fire on a threatened marine mammal in the nearshore ecosystem and fires that burn in watersheds adjacent to nearshore ecosystems impact the marine habitat by increasing nonpoint source pollutants above pre-fire levels. This research links the severity of wildfire to land cover changes that subsequently increase exports of pollutants and sediment to the nearshore environment. Not only is it replicable across other watersheds, it also indicates that terrestrial land management revolving around wildfire, including suppression, thinning, post-fire rehabilitation, and other activities changing land cover at a landscape scale, can be assessed for potential impacts to the nearshore environment. Coupling terrestrial and nearshore marine ecosystems in such a way may provide considerable insight to terrestrial impacts on the health and welfare of marine species at risk.

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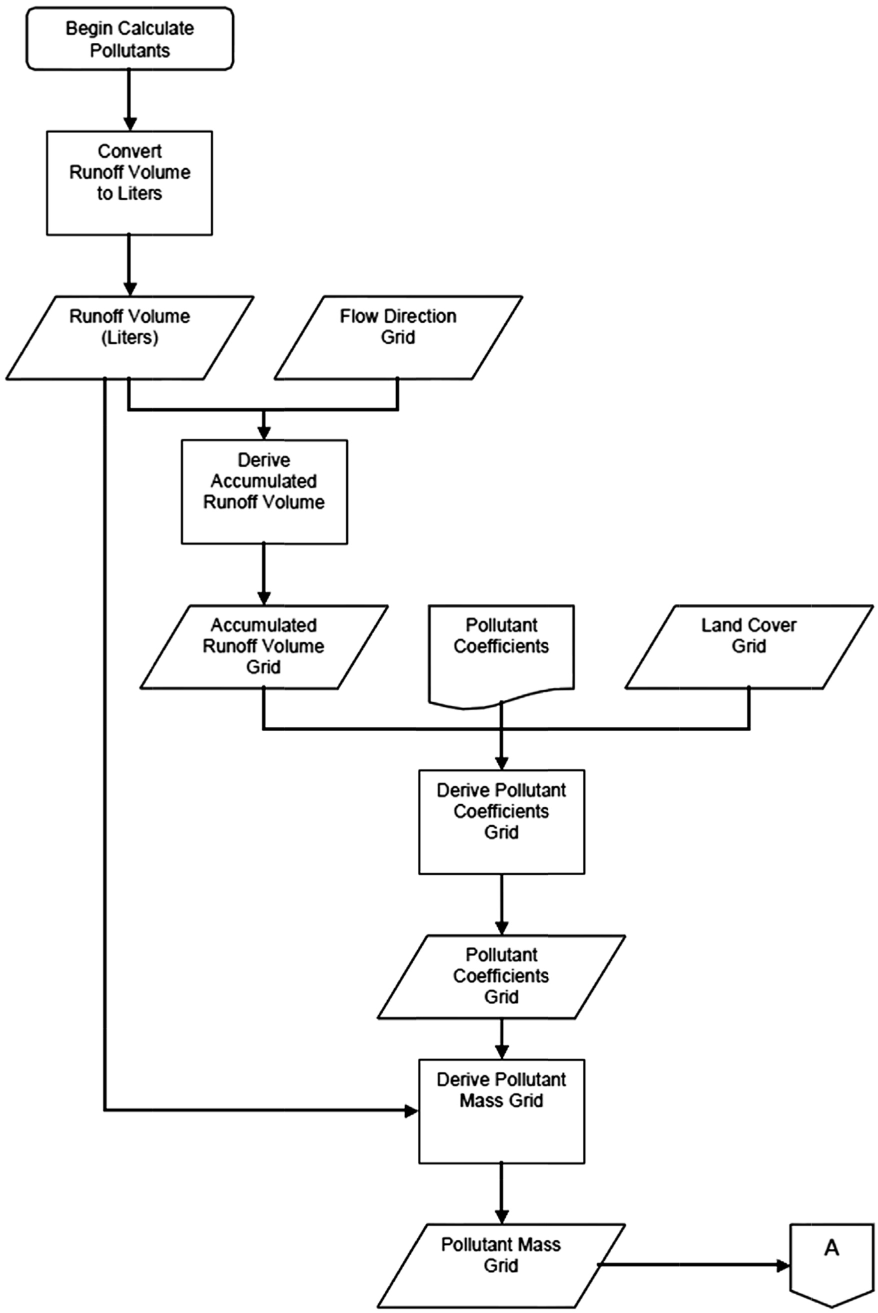
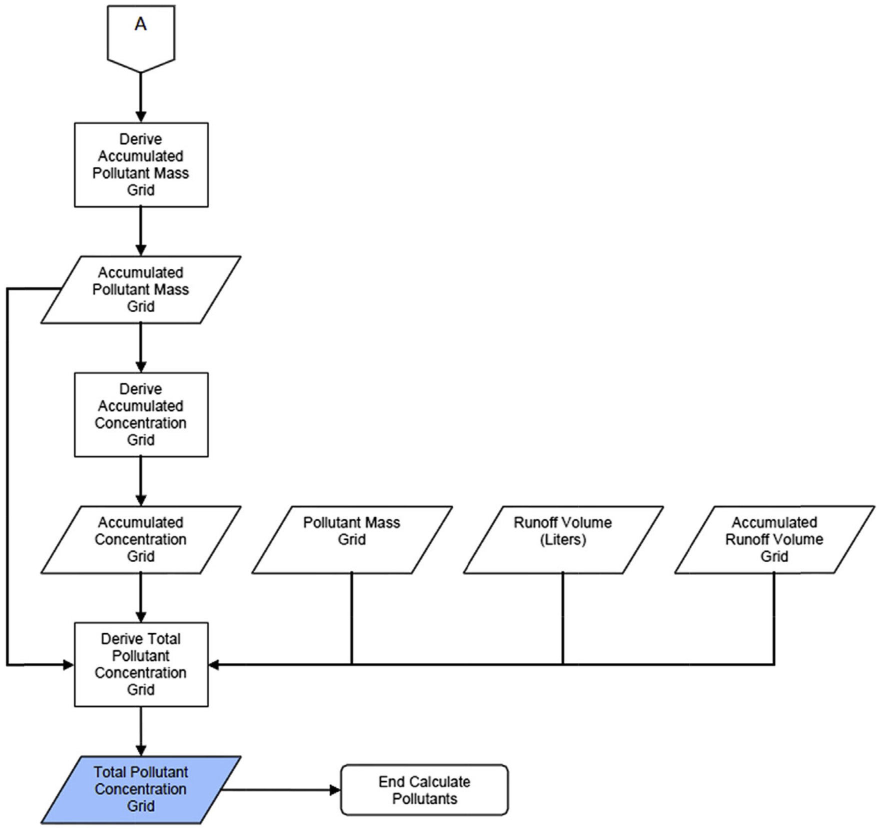
Appendix A. OpenNSPECT pollutant concentration estimation process (NOAA, 2012). Shading represents output dataset





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