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Distribution of terrestrial organic material in intertidal and nearshore marine sediment due to debris flow response efforts

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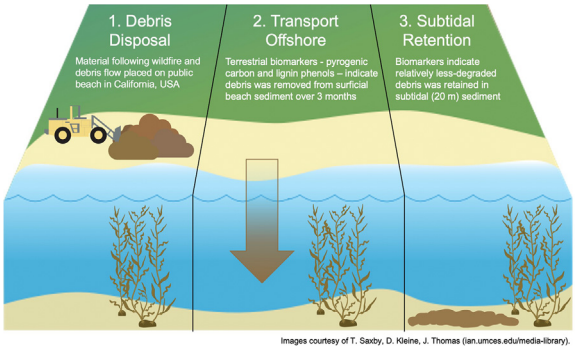
HIGHLIGHTS

- Terrestrial debris was deposited on a beach following a wildfire and debris flow.
- Lignin and pyrogenic carbon indicate debris was removed from beach sediment.
- Biomarkers indicate greater debris retention in deeper compared to shallower water.
- Biomarkers indicate greater debris degradation in shallower compared to deeper water.

GRAPHICAL ABSTRACT



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ABSTRACT

We examined the distribution and processing of terrestrial organic material, derived from the disposal of material from a massive debris flow event following a major wildfire in a coastal California (USA) catchment in intertidal and nearshore subtidal marine sediments. Organic matter biomarkers, pyrogenic carbon and lignin phenols, were used to trace the distribution of terrestrial debris material in marine environments. In intertidal sediments located <1 km east of the debris deposition site, pyrogenic carbon values did not significantly change and lambda values, a lignin measure, decreased over time, indicating little lateral transport of the disposed material. In subtidal sediment, pyrogenic carbon and lambda values were greatest in 20 m water depths indicating transport and deposition of this material nearshore. An additional lignin measure indicative of degradation suggested terrestrial organic material degradation in subtidal sediment decreased with distance from shore. Terrestrial biomarkers demonstrated that the disposed material was not detected in the top 20 cm of intertidal sediment but was retained in subtidal sediment offshore of the disposal site. Results suggest coastal management should incorporate consideration of the effects of debris disposal activities on nearshore benthic communities and biogeochemical cycling.

1. Introduction

Coastal regions around the globe are experiencing human population increases, with approximately 11 % of the world's population residing in coastal areas as of 2010 and coastal populations projected to increase by nearly 50 % by 2050 (Pörtner et al., 2019). While supporting increased human populations and their associated infrastructure, nearshore areas are at risk from events that are increasing in frequency and intensity due to climate change and sea-level rise, namely submergence, flooding, erosion, hurricanes, and inland salinization (Oppenheimer et al., 2019). The frequency of periods of drought followed by intense precipitation in Mediterranean regions, such as coastal California, is projected to increase over

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the next century (Swain et al., 2018). These events may increase the likelihood of sudden disturbances, such as wildfires and debris flows, that require rapid response to protect nearby communities. With climate disturbances projected to increase, there is a growing need to examine response and adaptation efforts in coastal areas, particularly those that occur under emergency scenarios.

In addition to increasing challenges posed by climate change, populated coastal regions are subject to management activities and anthropogenically-mediated disturbances, such as sediment addition or removal. Typically, sediment is dredged from waterways to maintain water depth or placed in new locations to mitigate coastal erosion (Martin, 2002). Less in known about the direct application of terrestrial sediment on beaches and in coastal ecosystems and its effects on coastal biogeochemistry. Nearshore permeable marine sediments are regions of high biogeochemical activity (Huettel et al., 2014), and a primary control of that activity is organic matter (OM) inputs (Boynton et al., 2018). Additional research is required to examine the effects of adding sediment, and as a consequence OM, in nearshore coastal ecosystems, particularly since anthropogenic disturbance in coastal regions may stimulate organic carbon mineralization rates and, by extension, reduce burial rates (van de Velde et al., 2018).

Coastal California is routinely subject to both the climate events and management activities described above. The State recently experienced a long drought, from 2012 to 2016, followed by a winter season with considerable rainfall in 2017 (Wang et al., 2017). In southern California, single storm events have the capacity to deliver half the annual suspended sediment load transported from the land to the ocean (Warrick et al., 2015), and the region is subject to frequent wildfires, which may cause further increases in suspended sediment concentrations in runoff (Aguilera and Melack, 2018; Coombs and Melack, 2013). Coastal regions in southern California are urbanized, and beaches are routinely mechanically managed (Schooler et al., 2019). The impacts of these compounding stressors are understudied in coastal beach ecosystems (Dugan et al., 2010).

Following the Thomas Fire in late 2017 (CAL FIRE, 2020) and the Montecito debris flow on January 9, 2018 (Kean et al., 2019), we investigated the distribution and processing of terrestrial OM following its emergency disposal on a local public beach (Goleta Beach, California). We examined how debris removed from Montecito, CA and disposed of on Goleta Beach was retained or distributed in sediments along the beach and in nearshore waters (20 m maximum water depth). The original debris sediment deposited on the beach was considered an endmember, and all intertidal (beach and estuarine) and subtidal marine sediment samples were analyzed for biomarkers of terrestrial OM, namely pyrogenic carbon and lignin phenols.

We hypothesized that the quantity of terrestrial OM would initially be high in intertidal sediment, and the content of this material would decline as it was reworked and swept into the nearshore waters due to the scouring and high wave energy typical of the winter season in the Santa Barbara Channel (Revell et al., 2011; Brzezinski et al., 2013). The aim of the debris disposal activities was to use the beach as a disposal site, but not to have the material retained in beach sediments. We further hypothesized that subtidal marine sediment collected immediately offshore of the disposal site in west Goleta Bay would have higher terrestrial OM content than marine sites located in east Goleta Bay. West Goleta Bay marine sites were located closest to the deposition site, and a previous study found that aquatic environments located nearest to terrestrial OM inputs displayed the highest terrestrial OM content in their sediment (i.e., streams) (Lowman et al., 2021). In contrast, east Goleta Bay is located offshore of the mouth of a nearby estuary that drained catchments not burned by the Thomas Fire. Unburned catchments export less suspended sediment following storms (Aguilera and Melack, 2018; Coombs and Melack, 2013). Therefore, due to the further distance from the deposition site and the upstream catchments being unburned, less sediment would reach east Goleta Bay sediment. Finally, we hypothesized that the terrestrial debris material placed on the beach would become significantly more degraded over time, and, as a result, the material detected in nearshore subtidal sediments would be similarly degraded. Lignin phenol degradation measures may be indicative of both abiotic and biotic degradation (Dittmar and Lara, 2001; Hedges and Ertel, 1982; Louchouarn et al., 1999). In particular, Lowman et al. (2021) found evidence of fresher terrestrial OM entering marine systems following subsequent storm events, suggesting these measures could also be used to interpret terrestrial OM degradation levels in aquatic environments. The overall goal of this study was to investigate the fate of terrestrial material added to coastal marine ecosystems after a major disturbance, specifically the human-mediated addition of significant quantities of terrestrial OM following a major wildfire and debris flow event.

2. Methods

2.1. Site description and event history

Goleta Beach (Longitude 119.828 W, Latitude 34.417 N) is a 1.6 km long sandy beach located in Santa Barbara County, California, USA, at the base of the Santa Ynez Mountains and along the northern, mainland coast of the Santa Barbara Channel. Goleta Beach is on the shoreline of Goleta Bay, a shallow bay at the mouth of the Goleta Slough that is protected from the prevailing northwest swell by a rocky headland. The watershed of Goleta Slough is 117 km^2, and seven creeks drain into the slough. The benthic habitats of Goleta Bay are dominated by soft sediments. The sandy beach is heavily used for recreation, and the pier is a popular fishing location. Goleta Beach lies in the large, longshore-transport dominated Santa Barbara littoral cell that ends at the Hueneme and Mugu submarine canyons in Ventura County (Patsch and Griggs, 2008). Although small coastal streams are the major source of sand for beaches, large quantities of sand, estimated to average 300,000 cubic yards/year based on Santa Barbara Harbor dredge records, move eastward along the Goleta to Santa Barbara coastline via longshore transport driven by the prevailing NW swell (Habel and Armstrong, 1978; Patsch and Griggs, 2006). Beaches of the Santa Barbara littoral cell, including Goleta Beach, exhibit considerable seasonal and interannual variation in profile and width (e.g., Revell et al., 2011; Revell and Griggs, 2006) rather than long-term trends. Episodic storms and El Niño-Southern Oscillation (ENSO) events strongly influence the sediment supply to these beaches and their resulting profile and condition (Barnard et al., 2009, 2017).

The region has a Mediterranean climate with cool, wet winters and warm, dry summers (Aguilera and Melack, 2018), and the marine ecosystems experience three distinct seasons - spring upwelling, summer stratification, and winter storm events with greater wave energy and episodic runoff November through March (Brzezinski et al., 2013). In nearshore regions (< 20 m), cross-shelf currents range from 0.01 to 0.02 m s^{-1} and along-shore currents reach 0.10 m s^{-1} (Fewings et al., 2015). The depth of closure (i.e., the depth beyond which the difference between seasonal and annual bathymetric changes are indistinguishable) ranges from approximately 2 m to 11 m water depth in Goleta Bay (Barnard et al., 2009).

On December 4, 2017, the Thomas Fire started and burned over 1100 km^2 of Santa Barbara and Ventura counties in California before being fully contained on January 12, 2018 (CAL FIRE, 2020). On January 9, 2018, a series of intense rainstorms (peak rainfall, 157 mm hr^{-1}) occurred in the Santa Ynez Mountains, triggering debris flows toward and in the town of Montecito and mobilizing approximately 680,000 m^3 of sediment (Kean et al., 2019). Following the debris flow event, response and rebuilding efforts included clearing sediment and debris from roads and properties in the region and transporting this material elsewhere. One of the locations chosen to receive this debris material was Goleta County Beach Park, located 25 km west of Montecito. Between January 16 and February 20, 2018, approximately 31,000 m^3 of material from the debris flow was deposited by the Santa

Barbara County Flood Control District on the west end of Goleta Beach and pushed as far beyond the low tide line as bulldozers could allow (Shank, personal communication, 2018, Appendix Fig. 1). In late April 2018, county officials returned to Goleta Beach to homogenize and redistribute a layer of organic-rich material that had congealed along the low tide line using bulldozers to push the material further offshore. Goleta Beach waters remained closed to recreation until July 6, 2018 due to elevated fecal indicator bacteria (*Enterococcus*) counts measured during weekly water testing (Santa Barbara County Public Health Department, 2018), and the results of another study found evidence of human waste in the disposal material, linking it directly to the deteriorated surf-zone water quality (Li et al., 2020).

2.2. Sampling design

In the spring of 2018, we collected sediment samples at three intertidal sites along Goleta Beach and at five subtidal sites in the nearshore region of Goleta Bay (Fig. 1, Appendix Table 1, Appendix Fig. 1). Intertidal samples were collected on an alongshore transect at increasing distances (650 m, 950 m) from the original disposal site on the west end of the beach. Due to truck and bulldozer activity, the disposal site (on February 2) and the low-tide line (on April 24) were sampled via shovel by Santa Barbara County employees (Appendix Fig. 1A). For all other intertidal sampling dates and locations, four replicate sediment cores were collected using 5 × 20 cm hand corers, stoppered on either end, placed in a cooler on ice, and transported immediately back to the laboratory at the University of California Santa Barbara (UCSB). On the last intertidal sampling date, the low-tide line at the disposal site was also sampled due to the formation of the organic-rich material layer; this site is not included in statistical analyses since it was sampled only once. Subtidal marine sediment samples were collected along two longitudinal transects, at 5 m, 10 m, and 20 m water depths perpendicularly offshore of the disposal site in west Goleta Bay and at 10 m and 20 m water depths offshore of the mouth of Goleta Slough in east Goleta Bay. West Goleta Bay was sampled as the "impacted" site, immediately offshore of the beach deposition site, and east Goleta Bay was sampled as the "control" site, receiving inputs of terrestrial material from the mouth of the Goleta Slough but located further from the deposition site. At each subtidal location, four replicate sediment cores were collected by SCUBA divers using the same 5 × 20 cm hand corers, stoppered on either end, and transported back to the laboratory in a cooler on ice. All samples were stored frozen (−20 °C) until sample processing. Intertidal cores were stored intact, since their size varied between 10 and 20 cm, and all subtidal cores were sectioned into 0–10 cm and 10–20 cm horizons prior to freezing.

2.3. Sample analyses

To prepare samples for analyses, two replicate cores from each site were selected, thawed, and 100 g of sediment from each core was weighed into clean aluminum tins. Samples were dried at 60 °C for 48 h, ground by hand using a mortar and pestle, and passed through a 2 mm sieve prior to being placed into combusted glass vials with a Teflon cap for storage.

Samples from each replicate core were analyzed for carbon stable isotope signatures, organic carbon (OC) content, pyrogenic carbon content, and lignin content. Isotopic and lignin analyses were used to determine the presence of terrestrial relative to marine organic matter (OM) in sediment samples, and the quantity of pyrogenic carbon (or charcoal) was indicative of material transported from the Thomas Fire burn scar or Montecito debris flow region and placed on Goleta Beach in February 2018.

Measurement of bulk stable organic carbon isotope signatures was done at the Department of Geological Sciences, University of Florida (UF) following Mays et al. (2017). To remove inorganic C, approximately 200 mg of sediment was weighed into in a container with ~250 mL 1 N HCl. After 48 h, the material was passed through a glass fiber filter with the aid of a vacuum pump. Decarbonated samples were then rinsed with deionized water to remove excess acid and chloride. The remaining sediment was oven-dried, separated from the filter, and stored in 20 mL glass scintillation vials. Ratios of ^{12}C and ^{13}C were then determined on a Thermo Finnigan Delta Plus XL isotope ratio mass spectrometer with a ConFlo III interface linked to a Costech ECS 4010 Elemental Combustion System. Carbon stable isotope results are reported as per mil (‰) in standard delta notation relative to Vienna PeeDee Belemnite (VPDB). Precision for $\delta^{13}C$ was ±0.21 ‰ based on nine analyses of UFCS, an internal laboratory standard. Total organic carbon (TOC) contents of the original sample after HCl acidification and in the digested samples were analyzed in duplicate (or additional times until <5 % relative error) on a Carlo-Erba NA-1500 CHS Elemental Analyzer.

Samples were also processed for pyrogenic carbon content using the Kurth-Mackenzie-Deluca method (Kurth et al., 2006), in which 1 g of sample was ground to <0.76 μm, and digested using 20 mL of 30 % peroxide (H_2O_2) and 10 mL of 1 M nitric acid (1 M HNO_3) at 100 °C for 16 h to remove non-charcoal carbon. After cooling and filtering with Whatman #2 filter paper, the sediment-laden filters were dried at 60 °C. Samples were carefully scraped from the filters, weighed, and analyzed for C content. It is assumed that all C that remained was pyrogenic carbon (Kurth et al., 2006). Pyrogenic carbon content (%PyC) was then calculated as the product of the C content in the digested sample and the ratio of pre- to post-digestion weight of the sample.

Lignin phenol analyses were used to determine the presence of terrestrial OM in sediment samples and the extent of diagenetic processing or degradation. A separate subsample from each replicate core was analyzed for lignin content using a modified alkaline cupric (CuO) oxidation method (Goñi and Montgomery, 2000) and a Varian 3800/Saturn 2000TM coupled gas chromatograph – mass spectrometer with a fused capillary column (DB-1 from J&W, 60 m, 320 μm) housed

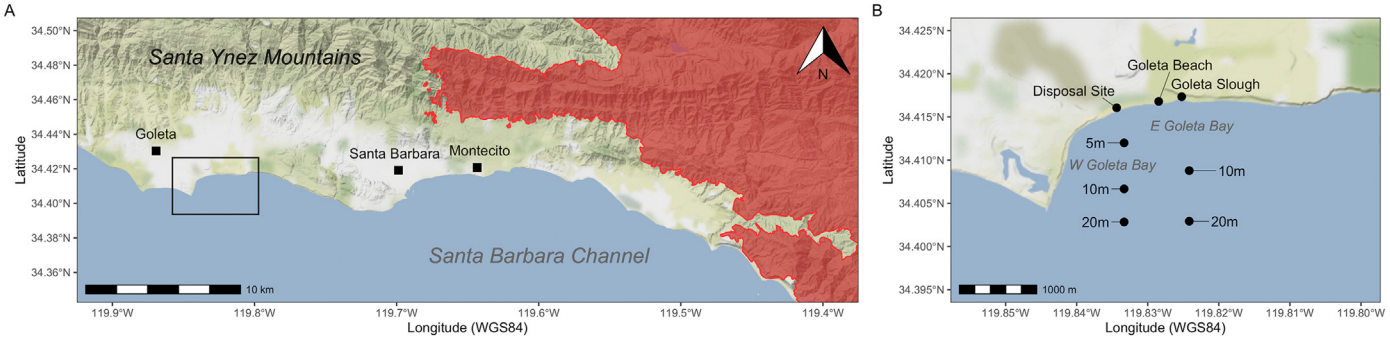

Fig. 1. (A) Map of the Santa Barbara region, with local municipalities denoted, and the Thomas Fire perimeter in red. The bounding box indicates the location of the inset. (B) Inset map with names of sampling sites along Goleta Beach and in Goleta Bay.

by Geotop at the Université du Québec à Montréal (UQAM) (Moingt et al., 2016). A standard reference material of estuarine sediment (SAG 05) was analyzed, and results were consistent with previously published values (Louchouarn et al., 2000; Moingt et al., 2016; Appendix Table 2). Although our use of a standard reference material suggested the extraction efficiency of the CuO method was consistent with past studies, calculating exact quantities of terrestrial OM in a sample using lignin phenol results is not advised. Rather, these results were used to assess relative inputs of terrestrial OM (Bélanger et al., 2017; Louchouarn et al., 1999; Moingt et al., 2016). The lignin phenols measured by the CuO oxidation method include syringyl, vanillyl, cinnamyl, and p-hydroxy phenols, in addition to another byproduct of the cupric oxidation of OM, 3,5-dihydroxybenzoic acid. These measures were used to calculate metrics of lignin amount and level of degradation. Lambda (Λ) values measure the sum of eight lignin oxidation products (i.e., all vanillyl, syringyl, and cinnamyl phenols), which, when normalized to organic carbon content within a sample, indicate the relative proportion of terrestrial OM to other OM in the system, for example marine primary production. Lignin phenol ratios were calculated to determine the degree to which the terrestrial OM present was processed or degraded. A decrease in the ratio of syringyl to vanillyl phenols (S/V) or the ratio of cinnamyl to vanillyl phenols (C/V) may be attributed to biodegradation processes (Dittmar and Lara, 2001; Opsahl and Benner, 1995). Inversely, an increase in both the ratio of *p*-hydroxyl phenols to vanillyl and syringyl phenols (P/(V + S)) as well as the ratio of 3,5-dihydroxybenzoic acid to vanillyl phenols (3,5-Bd/V) are interpreted as an increase in the level of degradation of the terrestrial source material (Dittmar and Lara, 2001; Hedges and Ertel, 1982; Louchouarn et al., 1999).

2.4. Data analyses

Data organization and analyses were performed using Microsoft Excel (version 16.24) and R Statistical Software (version 4.1.2, R Core Team, 2021). We performed data formatting and visualization using the *tidyverse* package (Wickham et al., 2019), the *sf* package (Pebesma, 2018), and the *ggmap* package (Kahle and Wickham, 2013) in RStudio (version 1.4.1106). Results are presented as mean values with one standard deviation. Prior to statistical analyses, pyrogenic carbon (%PyC), lambda (Λ), cinnamyl to vanillyl (C/V), *p*-hydroxyl phenols to vanillyl and syringyl phenols (P/(V + S)), and 3,5-dihydroxybenzoic acid to vanillyl phenols (3,5-Bd/V) values were log-transformed to meet the assumption of normal distribution.

To address our first hypothesis regarding transport of the terrestrial material along the beach, two linear mixed effects models were created for Λ and %PyC values including sampling date as a fixed effect and sampling site and replicate cores as nested random effects. We chose to use linear mixed effects models to incorporate the repeated sampling design at beach sites. Model creation and selection followed the protocol outlined by Zuur et al. (2009, Chapter 5), beginning with a linear model, adding fixed and random effects using a random intercept structure, examining additional variance structures, optimizing the fixed structure, and validating the best model fit using Akaike information criterion values alongside distribution of residuals. The *lme* and *glht* functions in the nlme and multcomp packages were used to create each model and compare categories within fixed effects (Hothorn et al., 2008; Pinheiro et al., 2021).

To address our remaining hypotheses regarding transport and processing of the disposed material in the nearshore marine environment, we constructed a series of multiple linear regressions for OM quantity measures Λ and %PyC as well as OM quality measures S/V, C/V, P/(V + S), and 3,5-Bd/V. All models included sampling site (west vs. east Goleta Bay), water depth, and sediment depth (0–10 cm and 10–20 cm horizons) as covariates and were constructed using the *lm* function in the stats package (R Core Team, 2021).

For all statistical tests, an alpha value of 0.05 was used to indicate significance unless otherwise noted. All code is publicly available at https://github.com/hlowman/coastal-debris-2022. Upon publication of the manuscript, data will be published on the Santa Barbara Coastal Long Term Ecological Research program data portal hosted by the Environmental Data Initiative (https://sbclter.msi.ucsb.edu/data/; Santa Barbara Coastal et al., 2022).

3. Results

Among all samples, organic content as %OC ranged from 0.05 % to 3.59 %, and mean values for intertidal (i.e., beach and estuarine) and subtidal marine sediments were 0.74 % ± 1.24 % and 0.40 % ± 0.15 %, respectively (Table 1). The $\delta^{13}C$ signatures of our samples ranged from −27 ‰ to −22 ‰, and mean values for intertidal and subtidal sediments were − 24 ‰ ± 2 ‰ and − 23 ‰ ± 1 ‰, respectively. Intertidal and subtidal sediment $\delta^{13}C$ signatures were not significantly different, and subtidal marine sediment cores collected from the western and eastern regions of Goleta Bay fell within the ranges of terrestrial particulate and kelp forest particulate OM; sediment sample $\delta^{13}C$ signatures ranged from −24 ‰ to −22 ‰. Based on these comparisons, stable C isotope measurements of our samples were not helpful in distinguishing inputs of debris disposal activities on Goleta Beach.

Pyrogenic carbon has no marine source and pyrogenic carbon content (%PyC) is another potential tracer of the fate of beach disposal deposits, and, specifically, can be used to assess the input of burnt terrestrial OM delivered with the debris from Montecito and the Thomas Fire burn scar. In our sediment samples, %PyC values ranged from 0.26 % to 4.75 %, and mean values for intertidal and subtidal sediments were 1.17 % ± 1.60 % and 0.82 % ± 0.36 %, respectively (Table 1). The results of a linear mixed effects model (LMEM) for intertidal sites (i.e., disposal site, Goleta Beach, and Goleta Slough) suggest that sampling date did not have a significant effect on overall %PyC content of intertidal sediments ($F(2,10) = 0.54$, $p = 0.60$), and except for the highest %PyC values detected at the beach disposal site on February 2 (4.67 % ± 0.12 %), %PyC values remained low at the other two intertidal sites on all three sampling dates (Fig. 2A). These results indicate little alongshore transport and redistribution of the debris material in an eastward direction.

At subtidal marine sites sampled on April 23, a multiple linear regression for %PyC content with sampling site, water depth, and sediment depth as predictor variables ($F(4, 15) = 7.59$, $p = 0.001$, $R^2 = 0.58$) indicated that both sampling site (ANOVA, $F(1) = 12.85$, $p = 0.003$) and water depth (ANOVA, $F(2) = 7.56$, $p = 0.005$) had a significant effect on %PyC content. The results of Tukey's post-hoc tests suggest that %PyC content was significantly greater ($p = 0.0004$) in western Goleta Bay sediment (0.97 % ± 0.38 %) than in eastern Goleta Bay sediment (0.60 % ± 0.15 %), and sediment collected at 20 m water depth had significantly greater ($p = 0.006$) %PyC content (0.99 % ± 0.36 %) than sediment collected at 5 m water depth (0.68 % ± 0.25 %). Greater pyrogenic carbon concentrations were present in western Goleta Bay, offshore of the disposal site, and in deeper water (Fig. 3A), indicating that %PyC transport from the beach was not uniform.

For lignin analyses, Λ values ranged from 0.03 to 4.6 mg/100 mg OC, and mean Λ values for intertidal and subtidal marine sediments were 0.71 ± 0.96 mg/100 mg OC and 1.06 ± 1.08 mg/100 mg OC, respectively. The results of a linear mixed effects model for intertidal sites (i.e., disposal site, Goleta Beach, and Goleta Slough) suggests that sampling date had a significant effect on overall intertidal Λ values ($F(2, 5) = 6.90$, $p = 0.04$). The results of a Tukey's post-hoc test suggest intertidal sediments collected on February 2 had significantly higher ($p = 0.0006$) Λ values (1.12 ± 1.39 mg/100 mg OC) than those collected on April 24 (0.06 ± 0.03 mg/100 mg OC). Although low-tide line data were not included in figures or models due to the singular

Table 1
Summary of organic carbon (%OC) content, isotopic signature ($\delta^{13}C$), pyrogenic carbon (%PyC) content, and lignin phenol signature (Λ, S/V, C/V, P/(V + S), 3,5-Bd/V) results for the study sites and sample dates. Lambda (Λ) values are presented in mg sample per 100 mg organic carbon. The remaining values are ratios of syringyl to vanillyl phenols (S/V), cinnamyl to vanillyl phenols (C/V), *p*-hydroxyl phenols to vanillyl and syringyl phenols (P/(V + S)), and 3,5-dihydroxybenzoic acid to vanillyl phenols (3,5-Bd/V). All results are presented as mean values plus or minus a single standard deviation, unless a parameter was only detected in one sample, in which case that single value is reported. Note – NAs represent samples in which lignin phenol signatures were not detected in either replicate analyzed.

Date	Ecosystem	Site	Depth	%OC	$\delta^{13}C$	%PyC	Λ	S/V	C/V	P/(V + S)	3,5-Bd/V
2/2	Intertidal	Disposal Site		3.54 ± 0.07	−27 ‰ ± 0 ‰	4.67 ± 0.12	2.44 ± 1.38	2.35 ± 0.54	0.20 ± 0.05	0.15 ± 0.06	0.30 ± 0.12
		Goleta Beach		0.17 ± 0.06	−23 ‰ ± 0 ‰	0.40 ± 0.06	0.12 ± 0.06	4.43 ± 0.57	0.43	0.72 ± 0.31	2.73 ± 0.08
		Goleta Slough		0.19 ± 0.13	−24 ‰ ± 2 ‰	0.30 ± 0.03	0.48	3.26	0.04	0.14	0.43
2/23	Intertidal	Disposal Site		0.24 ± 0.04	−26 ‰ ± 0 ‰	0.56 ± 0.33	0.70 ± 0.20	2.34 ± 0.17	0.09 ± 0.02	0.15 ± 0.07	0.41 ± 0.05
		Goleta Beach		0.15 ± 0.01	−24 ‰ ± 0 ‰	0.43 ± 0.02	0.09 ± 0.03	7.62 ± 1.84	NA	0.31 ± 0.02	2.71
		Goleta Slough		0.11 ± 0.08	−23 ‰ ± 0 ‰	0.32 ± 0.06	0.07	NA	NA	0.24	6.24
4/23	Subtidal	W Goleta Bay	5 m	0.40 ± 0.05	−23 ‰ ± 1 ‰	0.68 ± 0.25	0.38 ± 0.17	2.43 ± 0.94	0.22 ± 0.18	0.30 ± 0.07	0.99 ± 0.07
			10 m	0.40 ± 0.11	−23 ‰ ± 1 ‰	0.94 ± 0.41	1.56 ± 2.04	2.45 ± 1.02	0.16 ± 0.12	0.26 ± 0.18	0.51 ± 0.11
			20 m	0.62 ± 0.06	−24 ‰ ± 0 ‰	1.30 ± 0.20	1.81 ± 1.02	1.76 ± 0.48	0.14 ± 0.04	0.10 ± 0.01	0.27 ± 0.04
		E Goleta Bay	10 m	0.22 ± 0.06	−22 ‰ ± 0 ‰	0.52 ± 0.12	0.58 ± 0.13	2.54 ± 0.77	0.22 ± 0.11	0.22 ± 0.08	0.56 ± 0.03
			20 m	0.37 ± 0.07	−23 ‰ ± 1 ‰	0.68 ± 0.14	0.99 ± 0.42	1.81 ± 0.66	0.15 ± 0.04	0.25 ± 0.13	0.36 ± 0.01
4/24	Intertidal	Disposal Site		0.11 ± 0.01	−22 ‰ ± 0 ‰	0.40 ± 0.00	0.05 ± 0.02	8.29	3.09	3.12 ± 3.66	7.32
		Goleta Beach		0.13 ± 0.05	−23 ‰ ± 0 ‰	0.50 ± 0.06	0.08	4.58	NA	0.76	6.27
		Goleta Slough		0.07 ± 0.00	−23 ‰ ± 0 ‰	0.29 ± 0.04	NA	NA	NA	NA	NA
		Low Tide Line		2.72 ± 0.04	−27 ‰ ± 0 ‰	3.86 ± 0.00	1.64 ± 0.01	2.29 ± 0.06	0.27 ± 0.04	0.15 ± 0.00	0.50 ± 0.02

nature of sampling, Λ values for the sediment collected at the Goleta Beach low-tide line on April 24 were several orders of magnitude greater (1.64 ± 0.01 mg/100 mg OC) than other intertidal sediments collected that day (Fig. 2B). Values of Λ declined through time at all three intertidal sites across the three sampling dates (Fig. 2B). These results further support the indication of little alongshore transport and redistribution of the debris material to the east of the disposal site.

At subtidal marine sites sampled on April 23, a multiple linear regression for Λ values with sampling site, water depth, and sediment depth as predictor variables ($F(4, 15) = 3.01$, $p = 0.05$, $R^2 = 0.30$) indicated water depth had a significant effect on Λ values (ANOVA, $F(2) = 5.91$, $p = 0.01$). The results of a Tukey's post-hoc test suggest subtidal marine sediments collected at 20 m water depth had significantly higher ($p = 0.01$) Λ values (1.40 ± 0.84 mg/100 mg OC) than those collected at 5 m water depth (0.38 ± 0.17 mg/100 mg OC). Greater Λ values were detected in deeper water (Fig. 3B), again indicating that transport of the debris from the beach was not uniform.

Measures of S/V, C/V, P/(V + S), and 3,5-Bd/V were at or below the limit of detection in many of the intertidal sediment samples (Table 1), so we focused on the lignin measures indicative of processing in subtidal marine sediment samples (Fig. 4). Since subtidal marine sites were sampled only once, we examined how sampling location, water depth, and sediment depth affected lignin phenol signatures. Subtidal S/V values ranged from 0.93 to 3.88 with a mean value of 2.20 ± 0.79, and subtidal C/V values ranged from 0.05 to 0.48 with a mean value of 0.18 ± 0.10. In addition, subtidal P/(V + S) values ranged from 0.04 to 0.44 with a mean value of 0.22 ± 0.12, and subtidal 3,5-Bd/V values ranged from 0.11 to 1.35 with a mean value of 0.54 ± 0.33. Based on the results of a series of multiple linear regressions for S/V ($F(4, 15) = 0.90$, $p = 0.49$, $R^2 = -0.02$), C/V ($F(4, 15) = 0.80$, $p = 0.54$, $R^2 = -0.04$), and P/(V + S) ($F(4, 15) = 1.41$, $p = 0.28$, $R^2 = 0.08$) values, sampling location, water depth, and sediment depth did not have a significant effect on these lignin ratios. However, a multiple linear regression for 3,5-Bd/V values with sampling site, water depth, and sediment depth as predictor variables ($F(4, 15) = 3.83$, $p = 0.02$, $R^2 = 0.37$) indicated water depth did have a significant effect (ANOVA, $F(2) = 7.52$, $p = 0.005$).

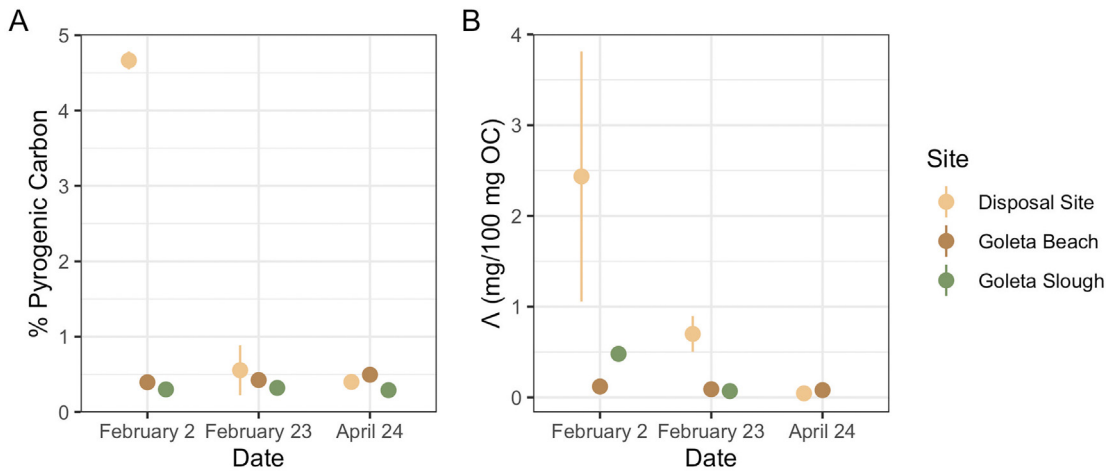

Fig. 2. (A) Mean pyrogenic carbon content and (B) Lambda (Λ) values for samples collected at intertidal sites (i.e., beach and estuarine) across all sampling dates in 2018. Note – the sediment collected at the Goleta Slough on April 24, 2018 had such low organic carbon content (0.07 %) that lignin phenols could not be detected.

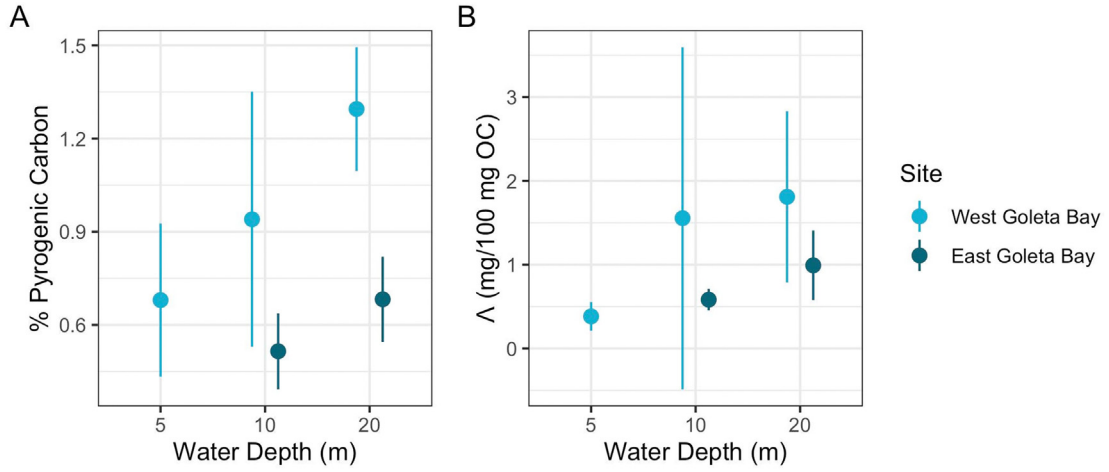

Fig. 3. (A) Mean pyrogenic carbon content and (B) Lambda (Λ) values for subtidal marine sediment samples collected across a range of water depths in Goleta Bay on April 23, 2018.

The results of a Tukey's post-hoc test suggest nearshore marine sediments collected at both 20 m and 10 m water depth had significantly lower (p = 0.0004 and p = 0.049, respectively) 3,5-Bd/V values (0.32 ± 0.12 and 0.53 ± 0.24, respectively) than those collected at 5 m water depth (0.99 ± 0.31), suggesting greater terrestrial OM degradation in shallower, nearshore sediments.

4. Discussion

4.1. Main findings

Debris flow material containing terrestrial organic matter (OM) was initially placed on Goleta Beach by county officials from January to February 2018 with the intention that it would be reworked and swept offshore by high wave-energy events. Our results do indicate that terrestrial OM content in intertidal sediments declined over time with little redistribution of the debris material along the beach. No significant change in pyrogenic carbon content (Fig. 2A) but a significant decrease in lambda (Λ) values over time (Fig. 2B) was observed. Contrary to the goals of the debris disposal, lignin and pyrogenic carbon measures indicate that debris material was retained and processed in nearshore marine sediments adjacent to the disposal site. Pyrogenic carbon (%PyC) content of subtidal marine sediment was greater in west versus east Goleta Bay marine sediments (Fig. 3A), indicating transport was primarily to nearby subtidal marine zones. Lignin measures indicative of degradation (P/V + S, 3,5-Bd/V) were below the limit of detection in most intertidal samples. However, 3,5-Bd/V values were greatest in shallow subtidal marine sediments (Fig. 4B), suggesting degradation of the disposed material was likely greatest closer to shore (5 m water depth) than in deeper water. In summary, our results suggest both intertidal and subtidal marine sediment stemming from the debris disposal underwent significant changes in pyrogenic and terrestrial OM content and were sites of OM retention and degradation.

4.2. Redistribution of disposed material in beach and nearshore environments

Measures of pyrogenic and plant-derived terrestrial OM in intertidal sediments traced the OM deposited on Goleta Beach during debris disposal activities, and the significant transport, processing, and burial of this terrestrially-derived OM in subtidal sediment at multiple depths. Over the three-month sampling period, constant %PyC values

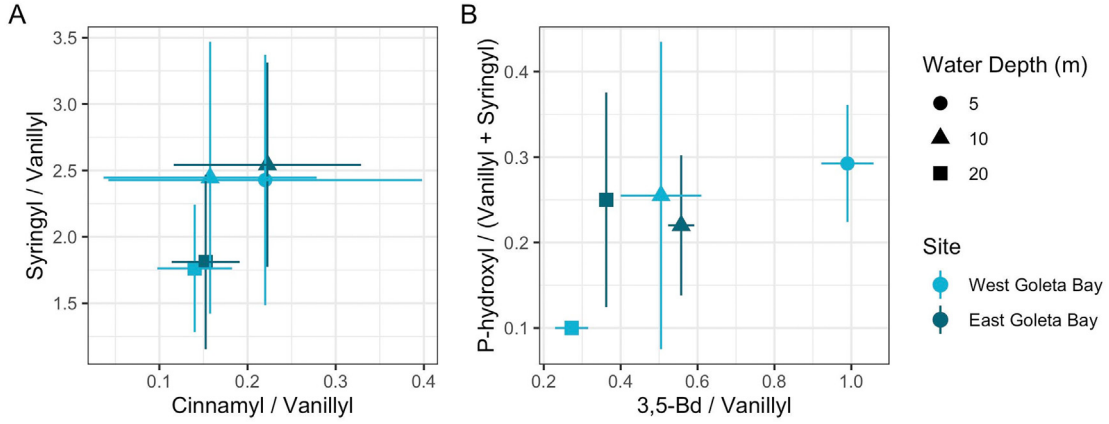

Fig. 4. Degradation measures of (A) syringyl vs. vanillyl phenols and cinnamyl vs. vanillyl phenols as well as (B) *p*-hydroxyl vs. syringyl and vanillyl phenols and 3,5-dihydroxybenzoic acid (Bd) vs. vanillyl phenols for subtidal marine sites sampled at different water depths.

in intertidal sediment suggested that appreciable longshore transport and redistribution of this material eastward along Goleta Beach did not occur, and declining Λ values suggested that the organic fine material was either buried by natural sand flux or removed from the sandy intertidal habitat of Goleta Beach. The beach surface underwent visible changes over the course of our study, transitioning from mostly dark soil to mostly sand and a greater prevalence of larger rocks and gravel that were likely part of the disposed material. In winter, the scouring of sand from Santa Barbara beaches is common (Revell et al., 2011), and the debris disposal event coincided with this scouring period as well as higher wave energy generated by winter storms in the region (Brzezinski et al., 2013; Shank, personal communication, 2018). In this region, transport of sand onto or away from shore happens relatively quickly (up to ±5 m of shoreline year^{-1}, Barnard et al., 2012), so the debris material placed on the beach may also have been rapidly buried by sand moving on and alongshore. Our results suggest that county managers successfully realized the goal of not retaining much of the finer, high organic content terrestrial material from the Montecito debris flows, at least in the surface sediments (0–20 cm) surrounding the debris disposal site on Goleta Beach.

Our sampling of subtidal sediment following the disposal event shows that the debris material was retained in nearshore marine sediment in Goleta Bay up to 20 m water depth for at least the three months following its disposal. East Goleta Bay, located at the mouth of the Goleta Slough and subject to routine delivery of storm runoff, was sampled as a "control" site in contrast to west Goleta Bay, which was immediately offshore of the beach deposition site. These sites were chosen for their differing exposure to terrestrial material inputs (i.e., west from debris deposition, east from storm runoff). Furthermore, the use of %PyC served as a biomarker specific to the debris flow material as the watersheds that drain into Goleta Slough and east Goleta Bay were unaffected by the Thomas Fire, and runoff from these regions would not be expected to contain similar levels of burnt terrestrial material. The contrast in results from these two areas of Goleta Bay indicate that the significantly greater %PyC measured in west Goleta Bay (LMEM $p = 0.003$, Fig. 3A) was a result of the debris flow material deposited on the beach, rather than input from the local watershed.

Due to poor water quality and beach closures, we were unable to perform multiple samplings in the nearshore marine regions to pair with intertidal samplings, and therefore cannot report on the progression of the presence of terrestrial OM in subtidal marine sediment as we have in intertidal sediment. However, for comparison, Λ values in subtidal sediments of east Goleta Bay in March 2017 were 1.44 ± 0.84 and 2.07 ± 0.65 in 10 m and 20 m water depths, respectively (Lowman et al., 2021). These values are a similar magnitude to the Λ values measured in west Goleta Bay, immediately offshore of the disposal site, during this study (Table 1). Since sampling for our study took place in a similar season and lignin endmembers may be interpreted regionally with a high degree of confidence (Prahl et al., 1994), the Λ values measured here suggest that the amount of terrestrial OM loading following the debris disposal event mimicked that of a typical winter storm event. Consequently, we suggest that the terrestrial OM deposited offshore in west Goleta Bay is within the range of the quantity delivered by storm events via Goleta Slough into east Goleta Bay. The Λ values measured in nearshore sediment suggest that terrestrial material from the Montecito debris flows was not transported far offshore, as county managers had intended, but rather was retained in sediment immediately offshore of the disposal site, in <20 m water depth and <1 km from Goleta Beach.

4.3. Potential impacts on biogeochemical cycling and nearshore ecosystems

Our results suggest that the debris deposited on Goleta Beach contained significant quantities of fresh terrestrial OM, as indicated by lignin analyses. Results from the study performed in the same region in 2016 and 2017 measured S/V and C/V values in suspended sediment collected from streams to be 2.45 ± 0.88 and 0.27 ± 0.13 (Lowman et al., 2021). These values are representative of minimally degraded terrestrial OM, delivered directly from the surrounding landscape into the stream, and may be interpreted in the context of our results because they are derived from the same system. We measured S/V and C/V values in beach sediment collected from the debris disposal site on February 2 to be 2.35 ± 0.54 and 0.20 ± 0.05, respectively (Table 1). These values correspond closely with the regional stream sediment lignin measures, leading us to conclude the debris deposited on Goleta Beach had been degraded relatively little. In another study performed in nearshore kelp forests, Santa Barbara Channel offshore particulate marine organic matter (OM) $\delta^{13}C$ signatures ranged from −22 ‰ to −19 ‰, and nearshore kelp forest particulate OM $\delta^{13}C$ signatures ranged from −23 ‰ to −17 ‰, while terrestrial OM inputs were indicated by $\delta^{13}C$ signatures ranging from −25 ‰ to −23 ‰ (Page et al., 2008). Several of our intertidal sediment samples had $\delta^{13}C$ signatures ranging from −27 ‰ to −25 ‰, suggesting input of material that was distinctly terrestrial in origin. Furthermore, %PyC measured in subtidal marine sediment offshore of the disposal site in west Goleta Bay suggested the composition of debris flow material differed relative to terrestrial material delivered from the watershed to marine sediments in east Goleta Bay (LMEM $p = 0.003$, Fig. 3A). As mentioned previously, the watersheds draining into east Goleta Bay were not burned by the Thomas Fire and would not be expected to yield similar levels of %PyC. Therefore, our results indicate that although the terrestrial OM placed on Goleta Beach may have mimicked the timing of seasonal storm runoff, its composition was more OM rich and labile than material typically delivered by storms and natural runoff processes.

In aquatic environments, frequent resuspension of nearshore sediment could cause increased abiotic degradation and lead to a less labile fraction of remaining OM in the sediment (Wainright and Hopkinson, 1997). In this study, 3,5-Bd/V values in subtidal marine sediment increased with decreasing distance from shore ($p = 0.02$, Fig. 4B). Typically, this pattern would be indicative of OM processing, but it should be accompanied by a similar pattern in P/(V + S) values (Dittmar and Lara, 2001; Hedges and Ertel, 1982; Louchouarn et al., 1999). Since a similar increase was not detected in P/(V + S) values (Fig. 4B), we propose that the trend in 3,5-Bd/V values was likely due to greater wave activity and resuspension in shallower subtidal areas (i.e., 5 m water depth) that led to increased degradation via physical fractionation processes rather than biogeochemical decomposition. Permeable marine sediments are demonstrated regions of high biogeochemical activity (Huettel et al., 2014) due to advective porewater exchange (Janssen et al., 2005) and high rates of OM resupply (Boynton et al., 2018). Due to the retention of the material in nearshore marine sediments shown here, further research is necessary to determine the extent to which the debris material may have been remineralized. In addition, organic detritus is one of the primary food sources for benthic communities (Woulds et al., 2016) suggesting future research on the effects of large additions of terrestrial OM on benthic consumers in subtidal sediments is warranted.

Less is known about the biogeochemical effects of pyrogenic carbon, or charcoal, addition to marine environments. This study provided an opportunity to use pyrogenic carbon as a tracer of the disposed material due to soil from the Montecito debris flows originating in the burn scar of the Thomas Fire. Though pyrogenic material may be either biotically or abiotically degraded to some extent (Abney and Berhe, 2018; Zimmerman and Gao, 2013), it is generally more refractory than terrestrial OM (Zimmerman, 2010) which means it could serve as a greater carbon sink in the coastal zone. The addition of pyrogenic carbon to marine sediments, along with its ability to sorb native OM and protect it from degradation (Zimmerman et al., 2011), may result in greater sediment OM content, but potentially of reduced accessibility to the benthos. Its ability to sorb or leach nutrients and other contaminants depends on the original vegetation and the temperature of

charring (Mukherjee and Zimmerman, 2013; Yao et al., 2012), which could also create environmental pollution concerns. In freshwater rivers, the addition of pyrogenic carbon has been demonstrated to increase dissolved organic carbon concentrations and alter dissolved organic matter composition and biofilm enzymatic activity (Thuile Bistarelli et al., 2021). Research regarding the direct impacts of pyrogenic OM on marine ecosystems remains scarce, but studies examining the immediate impacts of the Thomas Fire suggest its smoke plume caused significant deposition of black carbon in the Santa Barbara Channel (Wagner et al., 2021), as well as detectable shifts in surficial phytoplankton community composition (Kramer et al., 2020). Our study and others demonstrate the input of pyrogenic material in nearshore regions of the Santa Barbara Channel (Hunsinger et al., 2008; Mensing et al., 1999), and since the 2018 Thomas Fire, Santa Barbara County has experienced 12 additional wildfires (CAL FIRE, 2022). With regular fires occurring in the region and more intense and episodic precipitation events predicted in the coming decades (Swain et al., 2018), wildfire-debris flow events are likely to occur with increasing regularity. These events will deposit additional debris material, whether naturally or anthropogenically, on beaches, and we suggest future research should focus on the effects of episodic input events of burnt terrestrial material on coastal biogeochemistry and ecosystem function.

In addition to coastal marine biogeochemical considerations, the disposal activities on Goleta Beach very likely impacted the biotic communities of the sandy beach. Approximately 30 km east of Goleta Beach, in the Carpinteria Salt Marsh, deposition of the same Montecito debris material caused a decline in high marsh vegetation and a relative increase in mid marsh vegetation, highlighting the potential effects of the large tracts of bare soil temporarily created by the debris (Silva et al., 2022). Sediment additions to beaches, including such activities as beach nourishment and sediment disposal, have been demonstrated to significantly reduce species richness, abundance, and biodiversity of intertidal biota (Manning et al., 2014; Peterson et al., 2000; Schooler et al., 2019; Wooldridge et al., 2016). On Goleta Beach, and other nearby beaches, specialized guilds of invertebrates readily consume and remineralize giant kelp wrack that washes onshore (Lastra et al., 2008; Lowman et al., 2019), and these wrack-dependent biota have been shown to be particularly vulnerable to beach maintenance activities (Dugan et al., 2003; Schooler et al., 2019). Furthermore, the addition of dredged sediment with grain sizes that are too fine has been shown to inhibit burrowing by the dominant beach invertebrates of the region, with implications for long-term impacts to survival (Viola et al., 2014). Due to these known impacts on sandy beach communities, we encourage future studies to consider how the physical disturbance and lasting signal in grain size from sediment disposal activities may affect coastal ecosystems.

4.4. Conclusions and future directions

In this study, we used biomarkers of terrestrial OM as proxies for human disturbance and alteration of a coastal ecosystem, an approach that has also been used to detect agricultural (Eckard et al., 2017), deforestation (Bélanger et al., 2017), and industrial activity (Louchouarn et al., 1999). Although the large quantity of debris deposited on Goleta Beach was intended to be transported far offshore, we found evidence of significant retention of terrestrial OM in nearshore sediment (<20 m water depth) and inside the depth of closure. These findings suggest future debris disposal activities in the region should consider OM loading and lability as well as the overall quantity of material designated for disposal efforts. Nearshore benthic sediments are demonstrated regions of high biogeochemical activity (Huettel et al., 2014). However, the time and energy required for a soft sediment system to respond and recover may vary considerably between disturbances that are part of natural cycles (e.g., winter storm runoff) versus those that are mediated by human activity (e.g., sediment disposal) (Lugo, 2018). Furthermore, it is necessary to examine the effects of human activities intended for societal recovery and resilience and ensure they do not compound the potential negative consequences of the extreme climate events they are designed to address (Anderson et al., 2018).

Increased monitoring efforts immediately before, during, and after emergency response activities are critical for evaluating the effects and outcomes of these anthropogenically-mediated disturbance events. Regardless of the intended objective of the activity, real-time monitoring also allows for discovery of potential side effects and yields data tailored to inform future planning and environmental management efforts. Debris flows are the kind of sudden event that merit emergency response, particularly in a region as populous as southern California. These include more destructive events such as the 1982 Love Creek debris flow (Ellen and Wieczorek, 1988) and the 2018 Montecito debris flow described here, as well as smaller events like the debris flows in Leach Canyon following the 2018 Holy Fire (Guilinger et al., 2020) and in Boulder Creek following the 2020 CZU Lightning Complex Fire (Santa Cruz County, 2022). The frequency of coupled drought-intense precipitation events is projected to increase in California (Swain et al., 2018), which may lead to increased couplings of destructive wildfire-debris flow events. As the population grows in affected regions, this will also necessitate rapid response efforts that balance protection of infrastructure and ecosystems. Our results highlight the need for greater consideration of physical, biogeochemical, and ecological effects of emergency response activities, specifically debris disposal on beaches following wildfires and debris flow events.

CRediT authorship contribution statement

H.E. Lowman: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Resources, Data curation, Writing – original draft, Writing – review & editing, Visualization, Supervision, Project administration, Funding acquisition. **M. Moingt:** Methodology, Validation, Investigation, Resources, Data curation, Writing – review & editing. **A.R. Zimmerman:** Methodology, Validation, Investigation, Resources, Data curation, Writing – review & editing. **J.E. Dugan:** Resources, Writing – review & editing, Funding acquisition. **J.M. Melack:** Conceptualization, Resources, Writing – review & editing, Supervision, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A

Appendix Fig. 1. Images of our sampling sites during and immediately following debris disposal. The disposal site on Goleta Beach is pictured (A) during active debris disposal on February 16, 2018 facing west and (B) immediately following disposal on February 23, 2018 facing northeast. The remaining sampling sites are (C) on Goleta Beach immediately east of the Goleta Pier facing south and (D) in Goleta Slough facing northeast, both pictured on February 23, 2018.

Appendix B. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2022.156886.

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